PNNL-21477 FINAL

Columbia Estuary Ecosystem Restoration Program

2012 SYNTHESIS MEMORANDUM

FINAL

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January 2013

Report Documentation Page

Form Approved
OMB No. 0704-0188

1. REPORT DATE **JAN 2013**	2. REPORT TYPE	3. DATES COVERED **00-00-2013 to 00-00-2013**

4. TITLE AND SUBTITLE **Columbia Estuary Ecosystem Restoration Program. 2012 Synthesis Memorandum.**	5a. CONTRACT NUMBER 5b. GRANT NUMBER 5c. PROGRAM ELEMENT NUMBER
6. AUTHOR(S)	5d. PROJECT NUMBER 5e. TASK NUMBER 5f. WORK UNIT NUMBER
7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES) **Pacific Northwest National Laboratory,902 Battelle Boulevard,Richland,WA,99352**	8. PERFORMING ORGANIZATION REPORT NUMBER
9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES)	10. SPONSOR/MONITOR’S ACRONYM(S) 11. SPONSOR/MONITOR’S REPORT NUMBER(S)

12. DISTRIBUTION/AVAILABILITY STATEMENT
Approved for public release; distribution unlimited

13. SUPPLEMENTARY NOTES

14. ABSTRACT

15. SUBJECT TERMS

16. SECURITY CLASSIFICATION OF:			17. LIMITATION OF ABSTRACT	18. NUMBER OF PAGES	19a. NAME OF RESPONSIBLE PERSON
a. REPORT **unclassified**	b. ABSTRACT **unclassified**	c. THIS PAGE **unclassified**	**Same as Report (SAR)**	**117**	

Standard Form 298 (Rev. 8-98)
Prescribed by ANSI Std Z39-18

Executive Summary

The Bonneville Power Administration and the U.S. Army Corps of Engineers, Portland District (BPA/Corps) jointly instituted the Columbia Estuary Ecosystem Restoration Program (CEERP) to implement federal ecosystem restoration actions and research, monitoring, and evaluation (RME) in the lower Columbia River and estuary (LCRE). The CEERP is composed of three programmatic elements: the Strategy Report, Action Plan, and this Synthesis Memorandum. The overall goal of the CEERP is to understand, conserve, and restore ecosystems in the LCRE. Relative to this goal, the specific CEERP objectives are as follows:

- Increase the opportunity for access by aquatic organisms to shallow-water habitats.
- Increase the capacity and quality of estuarine and tidal-fluvial ecosystems.
- Improve ecosystem realized functions.

As a companion to the Strategy Report and Action Plan under the adaptive management process, this CEERP 2012 Synthesis Memorandum summarizes the state of the science of salmon ecology and habitat restoration in tidally influenced areas of the LCRE to provide an integrated scientific basis for the future strategic direction of ecosystem restoration. The findings and recommendations in this report are directed at refining both the action plan and the strategy for meeting the objectives of CEERP.

Methods

Development of this Synthesis Memorandum included the review of RME studies performed throughout the 234-rkm expanse of the tidally influenced main-stem and lower tributary sites below Bonneville Dam, as well as the principal peripheral embayments (Grays Bay, Youngs Bay, and Baker Bay). Spatial referencing was based on eight hydrogeomorphic reaches, A–H, which extend from the mouth of the Columbia River to Bonneville Dam. Information was acquired from peer-reviewed journal articles and completed contract reports initiated during the period from 1990 through mid-2012. The focus was on shallow-water habitats that are the primary sites for ecosystem restoration projects and recovery of juvenile salmon; however, a portion of the review took a more holistic approach to evaluating the LCRE ecosystem.

Conclusions

The CEERP objectives provided the bounds and focus of our review. To address this focus, we reviewed and synthesized information under four research questions. We summarize our conclusions associated with those four questions in the sections below.

What are the contemporary patterns of juvenile salmon habitat use in the estuary, and what factors or threats potentially limit salmon performance?

Patterns of estuary habitat use and the life histories of juvenile salmon are directly tied to their freshwater sources. Large releases of salmon from hatchery sources are a major driver of contemporary stock abundances and the arrival times, sizes, habitat preferences, and residence times of juveniles in the estuary. Because hatcheries target relatively few salmon stocks and phenotypes, the dominant estuary rearing behaviors today may or may not reflect the habitat and restoration needs of under-represented and

at-risk stocks. Furthermore, neither the interactions of hatchery- and natural-origin salmon nor the potential effects of hatchery releases on the estuary ecosystem have been investigated. It is unclear, for example, whether continued subsidies of similarly-sized hatchery smolts released in concentrated pulses during the spring have enhanced bird or other predator populations in the LCRE. Juvenile coho salmon are more prevalent in tidal wetlands within tributary systems than in main-stem sites. Many of the small juvenile salmon are wild spawned, and constitute a life history type not represented by the hatchery production system.

Habitat opportunity appears to be a major limitation to salmon performance. Many potential systems are simply unavailable due to migration barriers. Reduced flushing, leading to high-temperature and low-oxygen conditions, also appears to limit the time salmon can benefit from some wetland habitats during summer months. Tide gates, even those with “fish friendly” designs, improve access but are not as beneficial as more open hydraulic reconnections for either salmon movements or for maintenance of adequate water-quality parameters. Nonetheless, restoration activities that increase habitat opportunity are likely to benefit many salmon populations, and effort should be directed toward targeting sites that can be fully reconnected rather than left with restricted hydraulic connections

With regard to habitat capacity, the limited information about salmon performance in wetland sites indicate salmon are benefitting from wetland food production that results in relatively high growth rates. Wetland-derived insect prey also appears to be regularly transported to the wider ecosystem, where it is available to fish not inhabiting wetlands. However the overall loss of marshes in the LCRE and the reduction of a macrodetritus-based food web may have reduced the overall capacity of the system compared to historical capacities.

Competition and predation within wetlands requires more research but present data have not documented adverse effects on salmon performance. Additional research is needed, including potential direct or indirect interactions with non-native species. Predation studies have not been conducted in wetland sites, and bird predation in particular may be significant.

Do factors in the estuary limit recovery of at-risk salmon populations and evolutionarily significant units (ESUs)?

Estuary residency and habitat use vary among stocks and their associated entry locations, times, and sizes. These findings have important implications for selecting estuary restoration projects more strategically to satisfy the diverse estuary migration pathways and habitat requirements of salmon from different ESUs. However, despite a wealth of new data about stock-specific habitat use, life histories, and performance of juvenile salmon in the estuary, much remains to be learned about the importance of estuary rearing to population viability and salmon recovery. In the last decade, new tagging techniques, otolith chemical analyses, and an improved genetic baseline for Chinook salmon have greatly expanded our capabilities for interpreting stock-specific patterns of estuary rearing and migration. Genetic results have documented variations in the stock composition of Chinook salmon in various estuary reaches and habitats. Tagging studies and otolith chemical methods have described life history variations for a few genetic stock groups.

Most RME studies have evaluated salmon habitat use or performance within the estuary and have not determined whether estuary rearing conditions influence adult survival. New life-cycle approaches to research and monitoring are needed to quantify the estuary’s linkages to salmon populations and to

evaluate the importance of estuarine habitat opportunities for salmon recovery. A series of indicator populations and experimental methods should be employed to directly measure the contribution of estuarine habitats to adult returns and population viability.

Continued estuary monitoring is needed to more fully characterize juvenile life history variations within and among genetic stock groups, including at-risk stocks that are in low abundance and often poorly represented in estuary sample collections. Mid- and upper reaches (D – H) of the estuary have been surveyed less intensively than those in the lower estuary. Additional surveys will be required in this region to encompass the full range of habitat types or time periods for different genetic stock groups. Most RME studies have targeted shallow-water and near-shore areas, including habitat types that have been most intensively modified by historical development and that are the primary focus of estuary restoration. Methods for sampling deeper channels further from shore (e.g., purse seine, pair trawl, acoustic–tag monitoring, etc.) often select for high proportions of yearlings and hatchery fish that tend to move most rapidly through the estuary during punctuated migration periods. Additional surveys in deep channel habitats may be useful if the objective is to estimate survivals or migration rates for rapidly migrating stocks (e.g., chum, steelhead, sockeye) or to compare stock-specific life histories (i.e., subyearling and yearling migrants) across a wider range of estuarine habitat types.

Are estuary restoration actions improving the performance of juvenile salmon in the estuary?

Restoration in the LCRE can offer positive benefits to juvenile salmon in terms of opportunity, capacity, and realized function. Several positive trends were observed in the studies we reviewed. Hydrologic reconnections can increase opportunity for fish to access restored sites, as noted at Crims, Kandoll Farm, and Ft. Columbia. In terms of evaluating capacity, improvements in water temperature were noted at Kandoll Farm and South Slough while improvements in prey production were noted at Crims Island. A positive benefit of realized function was observed at Crims Island by examining residence time.

The primary direct beneficiaries of restoration of main-stem wetland habitats will be small subyearling Chinook and chum salmon with smaller numbers of larger yearling Chinook salmon found in shallow areas. Restoration of main-stem wetland habitats also has indirect benefits to juvenile salmon through export of organic materials, nutrients, and prey resources from shallow-water to main-stem areas. In order to restore life history diversity to Columbia River salmon populations, it is critical to protect, restore, and enhance the wetland habitat upon which these fish depend.

Our answer to this important question was based on limited AE information throughout the entire lower river and estuary. Of the 42 aquatic restoration projects that have been completed in the LCRE since 2004, only a small fraction (n=9) included AE monitoring that addressed elements relevant to juvenile salmon ecology; i.e., opportunity, capacity, and realized function. In many cases, AE research lacked pre-restoration data, reference sites, and/or statistical analyses aimed at specifically evaluating response of monitored metrics within the context of restoration actions. Further, of the existing nine AE studies, most (seven) were conducted in the lower 90 rkm of the estuary, and thus provide limited spatial coverage over the entire system from which inferences can be drawn. While these limitations present significant challenges with respect to effectively evaluating salmon performance, we conclude that restoration actions are correlated with increased opportunity, capacity, and realized function which provide benefits to juvenile salmon in the lower river and estuary.

What is the status of the estuary? Are estuarine conditions improving, declining?

The physical changes, including floodplain development, dredging of the navigation channel and harbors, and flow regulation, significantly altered the historical geomorphic and ecological state of the LCRE prior to the CREDDP studies. However, the rate of physical alteration has apparently slowed compared to the late 19th and early 20th century. Physical changes are still occurring. The navigation channel was deepened (1−3 ft) early in the present century, and channel maintenance, including dredge material disposal in the estuary is conducted periodically. Pile dikes, designed to maintain the navigation channel location and depth, have resulted in deposition of sediments and, in some cases, the formation of shallow-water habitats.

The habitat complexes within the present floodplain form a highly altered mosaic compared to historical conditions. The biological communities and geomorphology of the system are structured by natural disturbances (e.g., floods), with evidence that the habitat mosaic shifts spatially when forced by hydrological conditions and other controlling factors.

Non-native species are abundant and dominate vegetation, plankton, fish, and benthos assemblages. Very few "historic" (i.e., late 1800s) wetland habitats remain in the system. The rate of introductions of non-native species may be decreasing, but this is difficult to discern. Data show an expansion of invasive, highly competitive, non-native species such as reed canarygrass.

There is a legacy of contamination in sediments. Contamination of water and sediment from persistent chemicals is increasing and is of significant concern.

Through alteration in river flow dynamics and volumes, increases in water temperature, and sea-level rise, climate change is expected to affect the ecological processes of shallow-water habitats, and the capacity of the habitats to support young salmon.

Restoration projects focused on floodplain habitats have increased over the past decade. These actions are showing immediate benefit to juvenile salmon by providing access to habitats as well as processes supportive of ecosystem services of benefit to the entire estuary. Further, natural breaching of levees and dikes has opened areas of former floodplain habitats. The land surface formerly behind the levees had obviously subsided and most sites remain dissimilar to nearby reference sites even after several decades. Hence, the full return of floodplain habitats to their historical state will be protracted, especially those dominated by tidal forested swamps. Yet, these systems will predictably continue to provide services during development phase. Emergent marsh habitats show large changes during the first four to seven years with full development to reference conditions predicted to be on the order of 75 years or more. As evidenced in historical natural breaches, estuarine riparian and tidal forested habitats can develop within several decades of reconnection, and do have intermediate stages that are contributing services to the system.

Even with focused floodplain habitat restoration, net ecosystem improvement is hampered by development activities such as road construction and resource extraction in tributary watersheds draining into the lower floodplain habitats and broader LCRE. These upstream alterations can affect the rate and level of recovery of restoring habitats in the floodplain, as well as the resilience of these restored sites to periodic large-scale disturbances such as major flooding events and climate change.

Preface

The U.S. Army Corps of Engineers, Portland District (Corps) (Ref. No. AGRW66QKZ80031101) funded the development of this Synthesis Memorandum under agreements with the U.S. Department of Energy and the U.S. Department of Commerce for work by Pacific Northwest National Laboratory (PNNL) and the National Marine Fisheries Service (NMFS), respectively. The Synthesis Memorandum is one of three inter-related, annual CEERP (Columbia Estuary Ecosystem Restoration Program) deliverables; the others are the Strategy Report and Action Plan. The Synthesis Memorandum synthesizes the state of the science on salmon ecology in the lower Columbia River and estuary. It provides a scientific basis for the restoration strategies described in the Strategy Report, which in turn is used to implement restoration and research, monitoring, and evaluation actions outlined in the companion Action Plan. The actions are researched, monitored, and evaluated, and the results are synthesized in the next Synthesis Memorandum. The CEERP deliverables are intended to guide or inform, as appropriate, the Actions Agencies, the National Marine Fisheries Service, the Northwest Power and Conservation Council, restoration project sponsors, researchers, and various interested parties.

A suggested citation for this report is:

Thom RM, NK Sather, GC Roegner, and DL Bottom. 2013. Columbia Estuary Ecosystem Restoration Program. 2012 Synthesis Memorandum. Prepared by PNNL and NOAA Fisheries for the Portland District Army Corps of Engineers.

For more information about the study, please contact Ms. Cynthia A. Studebaker, the USACE's technical lead (503-808-4788).

Acknowledgments

We are grateful for the contributions made by many individuals throughout the preparation of this document. Dana Woodruff and Roy Kropp (PNNL) contributed the water properties and benthos sections, respectively. Lyndal Johnson (NOAA) contributed to the contaminates section. Catherine Corbett and Jina Sagar (Lower Columbia Estuary Partnership) provided information from reports from the ecosystem monitoring and reference site programs. Keith Marcoe (Lower Columbia Estuary Partnership) provided the analysis of habitat changes, helped obtained restoration project information, and fielded numerous questions along the way. A. Borde (PNNL) produced base maps for the report, and G. Johnson (PNNL) assisted in developing introductory material, and peer reviewed the draft. We thank the many people who contributed helpful comments that improved previous versions of the Synthesis document, including B. Ebberts and C. Studebaker (USACE), the Independent Science Advisory Board (ISAB). Susan Ennor and Mike Parker (PNNL) edited the report and Kathy Neiderhiser (PNNL) formatted the report.

Production of the report was funded by the Portland District Army Corps of Engineers.

Acronyms and Abbreviations

°C	degree(s) Centigrade (or Celsius)
AE	action effectiveness
BACI	before-after-control-impact
BAT	Beaver Army Terminal
BiOp	Biological Opinion
BPA	Bonneville Power Administration
C-CAP	Coastal Change Analysis Program
CEERP	Columbia Estuary Ecosystem Restoration Program
CREDDP	Columbia River Estuary Data Development Program
CREST	Columbia River Estuary Study Taskforce
Corps	U.S. Army Corps of Engineers
Council	Northwest Power and Conservation Council
CPUE	catch per unit of effort
CWT	coded-wire tag
7-DADM	seven-day average daily maximum
7-DAM	7-day mean maximum temperatures
d	day(s)
DO	dissolved oxygen
EM	emergent marsh
ESA	Endangered Species Act
ESU	evolutionarily significant unit
ETM	estuarine turbidity maximum
FCRPS	Federal Columbia River Power System
FL	fork length
h	hour(s)
HUC	Hydrologic Unit Code
ind/m^2	individuals per meter square
ind m^{-2} d^{-1}	individual per meter square per day
%IRI	percent index of relative importance
ISRP	Independent Scientific Review Panel
JBH	Julia Butler Hansen National Wildlife Refuge
km^2	square kilometer(s)
LCFRB	Lower Columbia Fish Recovery Board
LCRE	lower Columbia River and estuary
LCREP	Lower Columbia River Estuary Partnership
LWD	large woody debris

LOBO	Land-Ocean Biogeochemical Observatory
MA	management area
mg/L	milligram(s) per liter
mm	millimeter(s)
mm/d	millimeter(s) per day
MT/yr	metric tonne(s) per year
NOAA	National Oceanic and Atmospheric Administration
ODFW	Oregon Department of Fish and Wildlife
PAH	polycyclic aromatic hydrocarbon
PAR	photosynthetically active radiation
PBDE	polybrominated diphenyl ether
PCB	polychlorinated biphenyl
PIT	passive integrated transponder
RME	research, monitoring, and evaluation
rkm	river kilometer(s)
SD	standard deviation
USACE	U.S. Army Corps of Engineers (or Corps)
USGS	U.S. Geological Survey

Contents

Figures

Tables

1.0 Introduction

The Bonneville Power Administration and the U.S. Army Corps of Engineers, Portland District (BPA/Corps) jointly instituted the Columbia Estuary Ecosystem Restoration Program[1] (CEERP) to implement federal ecosystem restoration actions and research, monitoring, and evaluation (RME) in the lower Columbia River and estuary (LCRE). The BPA/Corps conduct the CEERP using an adaptive management process, which includes an Action Plan (BPA/Corps 2012) that contains the annual blueprint for ecosystem restoration and RME actions in tidally influenced areas of the LCRE floodplain.

As a companion to the Strategy Report and Action Plan under the adaptive management process, this CEERP 2012 Synthesis Memorandum summarizes the state of the science of salmon ecology and habitat restoration in tidally influenced areas of the LCRE to provide an integrated scientific basis for the future strategic direction of ecosystem restoration. Themes relevant to juvenile salmon were reviewed and synthesized under one of four research questions:

1. What are the contemporary patterns of juvenile salmon habitat use in the estuary, and what factors or threats potentially limit salmon performance?
2. Do factors in the estuary limit recovery of at-risk salmon populations and evolutionarily significant units (ESUs)?
3. Are estuary restoration actions improving the performance of juvenile salmon in the estuary?
4. What is the status of the estuary? Are estuarine conditions improving, declining?

The synthesis is based on a review of peer-reviewed and published literature regarding salmon ecology, restoration studies, and ecosystem ecology, conducted in the LCRE largely between 1990 and 2012, as well as selected relevant literature from other tidal river systems in the Pacific Northwest.

1.1 Columbia Estuary Ecosystem Restoration Program

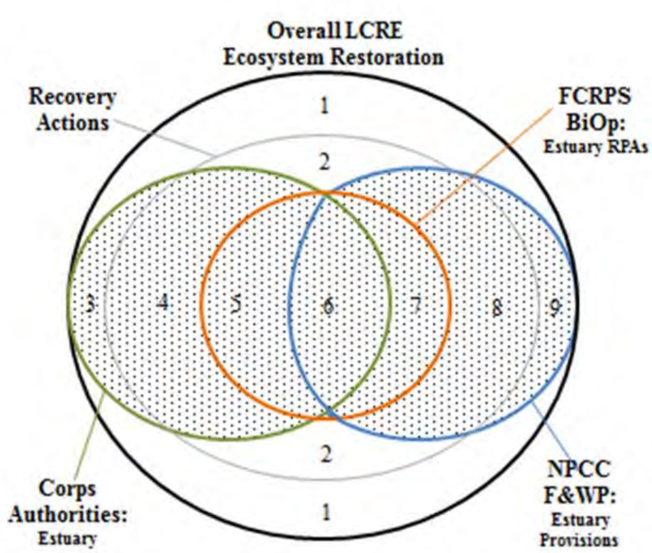
The CEERP is composed of three programmatic elements: the Strategy Report, Action Plan, and this Synthesis Memorandum. The 2012 Strategy Report describes the BPA/Corps fundamental strategy for implementing estuary habitat actions and RME. In addition, the CEERP is implementing Recommendation 3 of the Council's RME/Artificial Production Categorical Review Recommendation Report (ISRP 2010) to monitor and evaluate the effectiveness of habitat restoration actions in the LCRE. Finally, the Council's and Independent Scientific Review Panel's programmatic issues—i.e., the "lack of a clear synthesis or framework in the estuary linking habitat restoration actions to monitoring efforts to action effectiveness evaluations"—regarding the LCRE restoration effort (Council 2011) are intended to be addressed by the 2013 Strategy Report, the 2013 Action Plan (BPA/Corps 2012), and this 2012 Synthesis Memorandum.

[1] CEERP is an acronym coined in 2011 for the joint BPA/Corps efforts to restore LCRE ecosystems that started with the 2000 Federal Columbia River Power System (FCRPS) Biological Opinion (BiOp) (NMFS 2000) and now is responsive to subsequent FCRPS BiOps, the Northwest Power and Conservation Council's Fish and Wildlife Program, and various Corps restoration authorities.

Three primary drivers for the CEERP are as follows:

- Northwest Power and Conservation Council (Council) Fish and Wildlife Program (Council 2009) – the Council's program has strategies for estuary habitat reconnections, long-term effectiveness monitoring, estimation of juvenile salmon survival rates, impacts from estuary stressors, and partnerships.
- Water Resources Development Acts (Sections 206, 536, and 1135) and the Lower Columbia River Ecosystem Restoration General Investigations Study – the Corps has authorities to restore LCRE ecosystems under various federal laws.
- Biological Opinions (BiOps) for operation of the Federal Columbia River Power System (FCRPS) (NMFS 2000, 2004, 2008, 2010) – LCRE habitat restoration is an offsite mitigation action to help avoid jeopardizing Endangered Species Act (ESA)-listed salmonids with hydrosystem operations.

Note that the CEERP is one among many restoration programs presently operating in the LCRE. Others include those of the Oregon Department of Fish and Wildlife (ODFW), the Oregon Watershed Enhancement Board, the Lower Columbia Fish Recovery Board (LCFRB), the National Oceanic and Atmospheric Administration (NOAA) Restoration Center, the Washington Department of Fish and Wildlife.



Legend: LCRE restoration funded by…
1 = Entities besides the BPA and Corps for ecosystem restoration, e.g., CREST at Sharnelle Fee
2 = Entities besides the BPA and Corps for recovery actions for listed fish, e.g., LCFRB at Lower Washougal
3 = Corps outside BiOp and recovery, e.g., Johnson Creek springwater
4 = Corps outside BiOp, e.g., Tenasillahe Island
5 = Corps for BiOp, e.g., Post Office Lake
6 = BPA+Corps for BiOp, e.g., Col. Stock Ranch
7 = BPA for BiOp, e.g., Otter Point
8 = BPA outside BiOp, e.g., Duncan Creek
9 = BPA outside BiOp and recovery, e.g., Shillapoo wildlife mitigation

Figure 1.1. Nested Relationships Among CEERP Drivers and Overall LCRE Ecosystem Restoration. The shaded area represents the CEEERP. CREST is the Columbia River Estuary Study Taskforce. LCFRB is the Lower Columbia Fish Recovery Board.

1.1.1 CEERP Goal and Objectives

The CEERP is founded on a specific goal, principles, objectives, and management questions within a well-defined adaptive management process. As indicated previously, the overall goal of the CEERP is to understand, conserve, and restore ecosystems in the LCRE. The objectives of the CEERP reflect an ecosystem-based approach. They support and are consistent with the Council's estuary strategies[1] set forth in the 2009 Fish and Wildlife Program (Council 2009) and recommendations[2] from the 2010 Council RME/Artificial Production Categorical Review (ISRP 2010). The specific CEERP objectives are as follows:

- Increase the opportunity for access by aquatic organisms to shallow-water habitats.

  Habitat access/opportunity is a habitat assessment concept that "appraises the capability of juvenile salmon to access and benefit from the habitat's capacity" (cf. Simenstad and Cordell 2000). Elements for evaluating habitat opportunity include physical constraints to connectivity (migration barriers, water depth), and physiological limitations set by water-quality parameters (primarily temperature and dissolved oxygen).

- Increase the capacity and quality of estuarine and tidal-fluvial ecosystems.

  Habitat capacity is defined as the ability of a habitat to support functions benefiting salmon (Simenstad and Cornell 2000; Gray et al. 2002; Bottom et al. 2005). Positive factors defining habitat capacity include prey production and the resultant bioenergetic potential, while negative attributes include the presence and impacts of predators and competitors

- Improve ecosystem realized functions.

  Realized functions are a category of habitat assessment that includes direct measures of physiological or behavioral responses of fish to habitat opportunity and capacity that leads to increased performance (Simenstad and Cordell 2000; Gray et al. 2002; Bottom et al. 2005). Metrics defining performance are measures of fish benefit, such as diet and foraging success, residency and growth, condition, and life history diversity

1.1.2 The CEERP Adaptive Management Process

The CEERP adaptive management process, described in detail by Thom et al. (2011a), involves five phases (Figure 1.2): decisions, actions, monitoring/research, synthesis and evaluation, and strategy (Thom 2000). The CEERP proceeds through each of these phases adaptively informed by the results from the preceding phase(s). The adaptive management process allows adjustment in management decisions and actions over time, based on new scientific information in order to achieve long-term CEERP goals and objectives. As management questions are answered by RME results, program objectives and strategies are revised as necessary and inform future restoration and RME actions.

[1] Fish and Wildlife Program estuary strategies include habitat restoration work to reconnect ecosystem functions, long-term action effectiveness monitoring, evaluation of salmon and steelhead migration and survival rates, and evaluation of impacts from flow regulation, dredging, and water quality.

[2] A primary recommendation was, "The Council calls for the responsible entities to complete an estuary-wide synthesis prior to the initiation of the review of habitat actions."

Figure 1.2. CEERP Adaptive Management Process. Brown and blue boxes signify adaptive management phases and deliverables, respectively.

1.2 Memorandum Contents and Organization

The ensuing sections of this memorandum describe the approach used to synthesize the state of the science of salmon ecology, effectiveness of habitat restoration, and changes in the general ecosystem conditions in tidally influenced areas of the LCRE as of 2012 and present the overarching research questions (Section 2.0), under which research themes are organized in subsequent sections (3.0 through 6.0). Section 7.0 contains a summary of findings. Section 8.0 outlines recommendations. A list of literature cited in the narrative is provided in Section 9.0.

2.0 Synthesis Approach

Development of this Synthesis Memorandum included the review of RME studies performed throughout the 234-rkm expanse of the tidally influenced main-stem and lower tributary sites below Bonneville Dam, as well as the principal peripheral embayments (Grays Bay, Youngs Bay, and Baker Bay). Spatial referencing was based on eight hydrogeomorphic reaches, A through H, which extend from the mouth of the Columbia River to Bonneville Dam (Simenstad et al. 2011) (Figure 2.1, Table 2.1). The focus was on shallow-water habitats that are the primary sites for ecosystem restoration projects and their associated reference sites. Representative studies in the review covered a range of environments including forested swamps, scrub-shrub and emergent vegetation wetlands, backwater sloughs, and main-stem soft-sediment sites. Several studies directly examined the ecosystem response to hydrologic reconnections. A limited number of studies in the deeper water of the main channel were also included, where the interest has been salmon survival and migration rates, and not habitat improvement. Investigations initiated during the period from 1990 through mid-2012 were the primary focus of this review.

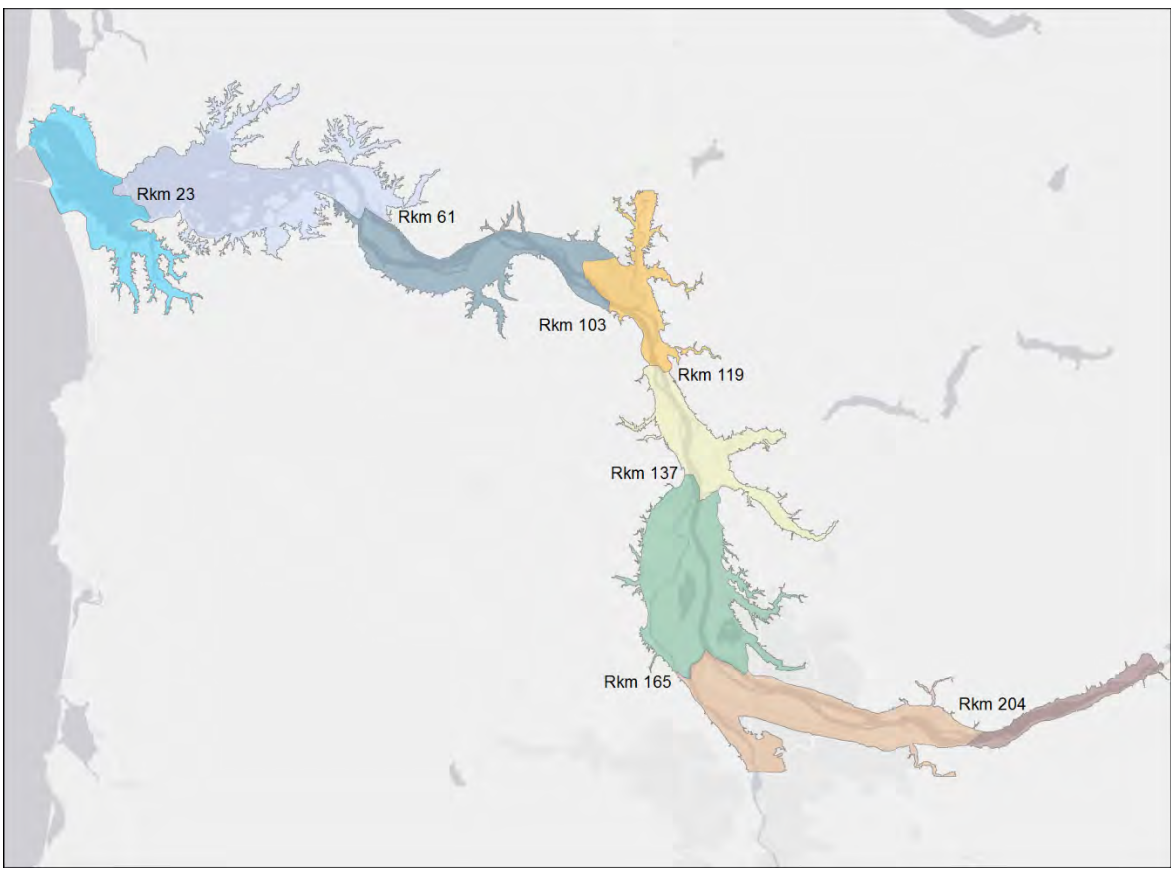


Figure 2.1. The Lower Columbia River and Estuary and Eight Hydrogeomorphic Reaches. River kilometers denote boundaries between reaches.

Table 2.1. Hydrogeomorphic Reach Names and Corresponding River Kilometer and River Mile Boundaries (Simenstad et al. 2011, Appendix A)

Reach	Rkm	R Mile
A	4–23	2–14
B	23–61	14–38
C	61–103	38–64
D	103–119	64–74
E	119–137	74–85
F	137–165	85–102.5
G	165–204	102.5–127
H	204–233	127–145

Information used in this synthesis was acquired from 1) peer-reviewed journal articles and completed contract reports; 2) electronic searches of the Web of Science and Aquatic Sciences and Fisheries Abstracts databases; and 3) BPA and Corps contract reports obtained online and via personal contacts. Where possible, the review used information found in completed summary reports in preference to annual reports. Many of the studies reviewed were multi-year studies, some of which were ongoing at the time of the review; future synthesis memoranda will integrate those continuing research efforts. The resulting synthesis represents a literature review-based effort, and not meta-analysis of data. The sections below expand the four overall research questions outlined in Section 1.0. For each research question, we examined literature sources and synthesized information primarily pertaining to aspects of juvenile salmon ecology in the LCRE. However, a portion of the review (i.e., Question 4) took a more holistic approach to evaluating the LCRE ecosystem by reviewing RME efforts that did not necessarily include a salmon-centric focus.

Definitions for juvenile life history stages used in this memorandum.

Diversity of salmon life histories has been described as an evolutionary strategy to spread risk and avoid brood failure in uncertain environments (Healey 1991; 2009). Major divisions in salmonid life history types include the subyearling and yearling rearing strategies, which refer to whether juveniles migrate to sea during their first year or reside for one or more years in lotic, riverine, tidal freshwater, or brackish environments (Myers et al. 2006). The subyearling life history stage can be further divided by size into fry (defined as fish ≤60 mm fork length) that can move rapidly to the ocean, or larger fish termed fingerlings that can rear in a variety of freshwater and brackish habitats (Bottom et al. 2005b). However, these subyearling stages are not discrete “types” per say; rather, diversity is represented by a continuum of juvenile residency patterns and adult spawning times that reflect spatial and temporal gradients in temperature during incubation and rearing (Brannon et al. 2004; Bottom et al. 2011). Even the yearling-subyearling migration designation is not necessarily fixed within a genetic stock of origin. In Chinook salmon, for example, both yearling and subyearling migrants are commonly produced by fall, spring, and summer runs of across the wide range of temperatures and elevations in the Columbia River Basin (Brannon et al. 2004; Copeland and Venditti 2009; Teel et al. 2009). The ability of many salmon stocks to express diverse life histories during the juvenile phase is thought to be an important adaptive mechanism for mitigating natural environmental variability (Healey 1991; Waples et al. 2009).

3.0 Contemporary Patterns of Juvenile Salmon Habitat Use in the Estuary and Factors that Potentially Limit Salmon Performance

Since 1990, substantial efforts have been made to understand the spatial and temporal distribution of juvenile salmon in habitats below Bonneville Dam. Studies have been divided into two main themes: 1) survival and migration timing of yearling and large subyearling salmon in main channel migration corridors, and 2) associations of salmon with the physical and ecological attributes of shallow-water environments (reviewed below). Survival and migration timing studies have primarily been concerned with effects of the Columbia River hydrosystem on ESUs that originate predominately above the Bonneville Dam, and recent research has brought many new technical advances to fish tracking and to improving the hydrosystem for fish passage. The second initiative has focused for the first time on smaller salmon that have a longer migration period and depend on shallow-water wetlands, sloughs, and intertidal beaches at which to reside and grow before entering the ocean. Studies in this initiative include monitoring of time-series as well as investigation of habitat associations in both "natural" habitats and areas undergoing restoration actions. In the ensuing sections, we concentrate on synthesizing information pertaining to salmon-habitat associations. We first review the species, life history types, and migration timing of salmon present in habitat studies, and summarize the spatial variation in hatchery versus unmarked fish. Then we summarize habitat associations and possible limiting factors in salmon requirements for habitat opportunity, habitat capacity, and salmon performance (Simenstad and Cordell 2000; Gray et al. 2002; Bottom et al. 2005).

3.1 Current Patterns of Salmon Species, Life history Types, and Migration in the LCRE

We identified 10 major studies since 2002 that have investigated juvenile salmon habitat associations (Table 3.1). Work has concentrated on shallow-water habitats at main-stem and wetland habitats in reaches A through H, including sites within peripheral bays in reaches A and B (Youngs Bay, Baker Bay, and Grays River). Six species of salmon and anadromous trout were identified in these shallow-water habitats: Chinook salmon (*Onchryrhchus tshawytscha*), coho salmon (*O. kisutch*), chum salmon (*O. keta*), sockeye salmon (*O. nerka*), steelhead (*O. mykiss*), and coastal cutthroat trout (*O. clarkii*). However, of these only Chinook, chum, and coho salmon were present in high abundance. These species and their various stocks display variations in juvenile life history characteristics and in the timing and pathways of their seaward migrations.

Yearling and subyearling life history stages are evident in Chinook and coho salmon, while chum salmon are primarily captured as fry migrants. Various tracking and monitoring studies indicate yearling Chinook and coho salmon and steelhead primarily use main channel migratory pathways during spring (e.g., Dawley et al. 1986; Magie et al. 2008; Weitkamp et al. 2012), although during winter, larger Chinook and coho salmon tagged in the Sandy River delta were found to have relatively long residence times of 24–34 d (G. Johnson et al. 2011; Sather et al. 2012). Large smolted subyearling Chinook salmon also tend to migrate rapidly through the lower river (Dawley et al. 1986; Harnish et al. 2012). However, a portion of these larger fish are also found in shallow-water habitats (e.g., Poirier et al. 2009a, b; Bottom et al. 2011; Sather et al. 2011; Roegner et al. 2012). In contrast, smaller subyearling Chinook and chum salmon make substantial use of shallow subtidal and intertidal habitats of diverse natures (discussed

Table 3.1. Relative Maximum Abundance of Juvenile Salmon Sampled From Shallow-Water Sites in the Lower Columbia River and Estuary. Abundance was categorized by catch per unit effort (Low, <10; Medium >10 – 100; High, >100) or density (Low < 0.01; Medium: >0.01 to 0.1; High >0.1 ind/m^2) X = species not found; WQ = water quality; HST = high summer temperatures recorded (>19°C); LDO = low dissolved oxygen recorded (<6.0 mg/L).

					Relative Abundance					
Reach	Site	Rkm	Habitat	Study	Chinook	Chum	Coho	Years	WQ	Citation
A	Ilwaco	6	Fringing wetland	Survey	L	H	X	2011		Sagar et al. 2012
A	Clatsop Beach	8	Main-stem	Survey	H	M	L	2002-2008,	HST	Roegner et al. 2008;
	West Sand Is	10	Island	Survey	M	H	L	2010-present	HST	Bottom et al, 2011;
	Pt. Adams Beach	20	Main-stem	Survey	H	H	L		HST	Roegner et al. 2012
	Pr Elice	22	Main-stem	Survey	H	H	L		HST	
A	Vera Slough(a)		Fringing Wetland	Restoration	L	L	L	2005-2010	HST	G. Johnson et al. 2009
A	Haven Is[(a)]		Wetland Island	Old Breach	L	L	X	2009	HST	CREST 2011b
A	Colewort Ck[(a)]		Fringing Wetland	Restoration	L	L	M	2010-present	HST	CREST 2012a
A	Walooski[(a)]		Fringing Wetland	Old Breach	L	X	X	2010	HST	G. Johnson et al. 2009
B	Russian Island	35	Wetland Island	Survey	L	L	L	2002-2008	HST	Bottom et al. 2011
	Seal Island	37	Wetland Island	Survey	L	L	L	2002-2003	HST	
	Karlson Island	42	Wetland Island	Survey	L	L	L	2002-2004	HST	
	Welsh Island	53	Wetland Island	Survey	H	L	L	2004-2005	HST	
B	Svensen	40	Fringing	Old Breach	L	X	X	2008	HST	Diefenderfer et al. 2012
	Karlson	42	Wetland Island	Old Breach	L	L	X	2008	HST	
	Miller Sands	45	Wetland Island	Restoration	L	L	X	2009	HST	
B	Julia Hansen Butler	55	Fringing Wetland	Restoration Control	L M	L L	L L	2007-2010	HST LDO	J. Johnson et al. 2011; 2012
B	Kandoll Farm[(b)] Johnson Farm[(b)]		Wetland Island	Restoration	L L	H M	M M	2005-2010	HST	Roegner et al. 2010; G. Johnson et al. 2011
B	Tenasillahe Island Welsh Island	60 53	Wetland Island	Restoration Control	L M	X L	X X	2006-2008	HST LDO	J. Johnson et al. 2011; 2012
B	L. Elochoman Slough	58	Main-stem	Survey	H	L	L	2002-2008	HST	Roegner et al. 2008;
	Upper Clifton Ch	59	Main-stem	Survey	H	L	L	2002-2008	HST	Bottom et al. 2011;
	E. Tenasillahe Is	61	Island	Survey	M	L	L	2002-2008	HST	Roegner et al. 2012

Table 3.1. (contd)

Reach	Site	Rkm	Habitat	Study	Relative Abundance			Years	WQ	Citation
					Chinook	Chum	Coho			
C	Wallace Island	77	Wetland Island	Survey	M	L	L	2006-2007	HST	Bottom et al. 2011
C	Crims Island	87	Wetland Island	Restoration	H	M	L	2004-2008	HST	Haskell and Tiffan 2011
C	Ryan Island	61	Wetland Island	Survey	M	L	X	2009	HST	Jones et al. 2010;
	White Island	72	Wetland Island	Survey	M	L	L	2009-2011	HST	Sagar et al. 2012
	Ryan Island	84	Wetland Island	Survey	L	L	X	2009	HST	
	Lord-Walker Is	99	Wetland Island	Survey	M	L	L	2009		
C	Lord Island	101	Wetland Island	Survey	M	L	L	2006-2007	HST	Bottom et al. 2011
D	Cottonwood Is	113	Main-stem Off Channel Wetland	Survey	H	L	L	2009-2010	HST	Diefenderfer et al. 2011
E	Burke Island	131	Wetland Island	Survey	L	X	X	2011	HST	Sagar et al. 2012
	Goat Island	131	Wetland Island	Survey	L	X	X	2011	HST	
	Deer Island	132	Wetland Island	Survey	M	X	L	2011	HST	
E	N. Deer Is Slough	132	Slough	Tide-gated	X	X	X	2009	HST	Poirier et al. 2009a
	S. Deer Is Slough		Slough		L	X	L		LDO	
	Tide Creek		Stream		X	X	M			
E	L. Willamette R		Main-stem	Survey	H	?	?	2001- 2003		Friesen et al. 2007
F	Campbell Slough	149	Wetland	Survey	M	L	L	2007-9,2011	HST	Jones et al. 2010; Sagar et al. 2012
G	Sandy River delta	~200	Main-stem Off Channel Wetland	Survey	H	M	L	2007-2012	HST	Sather et al. 2009, 2011; G. Johnson et al. 2011
H	Franz Lake	221	Wetland	Survey	M	L	L	2008-9, 2011	HST	Sagar et al. 2012

(a) Youngs Bay.
(b) Grays River.

below), and subyearling coho are often abundant in the lower sections of tributary rivers (Poirier et al. 2009a, b; Roegner et al. 2010; CREST 2012a). Hence, this synthesis concentrates on Chinook salmon with more limited assessments for chum and coho.

3.1.1 Chinook Salmon

Subyearling Chinook salmon are present in the LCRE year-round, make extensive use of shallow-water habitats, and appear to exit to the ocean in a variety of sizes ranging from fry (≤60 mm) to large, late autumn to winter migrants (Bottom et al. 2011; Roegner et al. 2012). Nearly every habitat investigated—isolated tide-gated wetlands, areas behind both new and old dike-breaches, beaches along the main-stem Columbia River, as well as relatively natural herbaceous and forested wetlands—has yielded Chinook salmon (Table 3.1). Restoration activities improving hydrological connections invariably find near-immediate increases in Chinook salmon abundance (Roegner et al. 2010; Haskell and Tiffan 2011; J. Johnson et al. 2008; 2011b). However, spatial and temporal variation are evident in fish density (or catch per unit of effort [CPUE]), size, and genetic stock of origin both among reaches and among habitats within reaches (detailed below). These variations reflect the particular uses of habitats by life history stages as well as the proximity of habitats to stocks of migrating fish. For example, restoration sites that lack a strong upstream source of migrants, such as Vera Slough in Youngs Bay (G. Johnson et al. 2007), have lower CPUEs and stock diversities than sites such as Cottonwood Island (reach C) that are available to a large number of ESUs (Diefenderfer et al. 2011). Densities vary widely but can exceed 1.0 individual per meter square (ind/m^2) (Table 3.1). Many intertidal wetland sites are dominated by fry-sized salmon, which are present at relatively high densities during spring and early summer, while larger individuals are often found contemporaneously in adjacent deeper channels (Bottom et al. 2011; Haskell and Tiffan 2011), or farther downstream (Roegner et al. 2012). Small, unsmolted subyearling Chinook salmon appear to be the dominant salmon species and life history type using shallow-water habitats. However, all Chinook salmon life history types have been identified in shallow water systems.

3.1.2 Chum Salmon

Juvenile chum salmon migrate primarily as fry (≤60 mm); they have a punctuated migration period extending from February through May or early June (Hillson 2009; Roegner et al. 2010, 2012; Sather et al. 2011). At main-stem sites, chum salmon are generally the second most abundant salmonid after subyearling Chinook salmon. In the Sandy River delta (reach G) during 2007−2010, chum salmon composed 0.5% of the total fish and 10% of the salmon populations, with a mean density in spring of ~0.01 ind/m^2 (Sather et al. 2011). Chum were moderately common at most wetland sites in Cathlamet Bay and other main-stem island complexes, composing between 0.01 and 0.08% of the total fish catch, although CPUE varied among years (Bottom et al. 2011). In the lower estuary, chum salmon were very abundant from February to May, peaking in April, and were absent thereafter (Bottom et al. 2011; Roegner et al. 2012). They composed 1.1% of the total fish population sampled from 2002 to 2007 but exhibited a strong spatial gradient, making up 2.0% of the total and 34.8% of the salmon population in the estuary versus 0.04% and 0.8%, respectively, in reach B. Densities were also much higher in the saline estuary (up to 1.0 ind/m^2) than contemporaneously at tidal freshwater stations in reach B (Bottom et al. 2011). However, chum salmon were not abundant at all habitats during their migration window. At Crims Island (reach C), Haskell and Tiffan (2011) sampled just 221 chum salmon over a 4-year period. Few or no chum were reported from Deer Island (Poirier et al. 2009a), in tide-gated sloughs at Julia Butler Hansen National Wildlife Refuge (J. Johnson et al. 2009, 2011), or within tidal channels at

Cottonwood Island (Diefenderfer et al. 2011). At all sites, the majority of chum salmon were fry migrants, but in reach A ~10% were fingerling-sized (Roegner et al. 2012).

High abundances of chum salmon were present in restoration sites in Grays River (Roegner et al. 2010; G. Johnson et al. 2012), which remains one of the few consistent natural chum spawning habitats (in addition to the existing hatchery program). More than 3000 chum salmon per tide were sampled from tidal creeks in Kandoll Farm during 2010 (G. Johnson et al. 2012). Chum salmon vacate the Grays River intertidal habitats by the beginning of May and apparently move rapidly to the estuary (Bottom et al. 2009; Roegner et al. 2010). Other major chum salmon spawning areas in the Columbia River include Duncan Creek, Washington (reach H), and the Pierce–Ives Island complex below Bonneville Dam (reach H), where fry emergence occurs over an approximately 25-d period ranging from March to April (Tomaro et al. 2007; Hillson 2009). The presence of chum salmon in Colewort Creek (Youngs Bay) suggests the presence of undetected spawning is occurring in the Lewis and Clark watershed (CREST 2012a).

3.1.3 Coho Salmon

Subyearling coho were relatively uncommon at most Columbia River wetland and main-stem sample sites (Table 3.1), but were found in tidal freshwater tributaries, e.g., Grays River (Roegner et al. 2010), Tide Creek (Poirier et al. 2009a), Colewort Creek (CREST 2012a), and Franz Lake sites in reach H (Jones et al. 2010). Coho salmon occurred infrequently in wetland channels, and their total abundance remained relatively low at all sites except for the mixed wetland site at Lord Island in 2007 (Bottom et al. 2011). In the Sandy River delta, coho salmon composed 8% of the salmon population but were at relatively low densities (<0.01 ind/m^2). At lower river and estuary main-stem sites, Roegner et al. (2012) caught only 202 coho from 2002 to 2007, most of which were yearlings. In main channel sites, coho yearling migration occurred from late April through May during 2007-2010, and abundances during peak travel time were relatively high (Weitkamp et al. 2012). There was little evidence of “nomads” in the main-stem Columbia River (Koski 2009).

3.1.4 Other Salmon and Trout

While other species of salmon and trout (e.g., sockeye salmon, steelhead, cutthroat trout) have been captured at various habitats sampled in the LCRE, catches appear to be incidental at most shallow-water sites. It is unknown whether this reflects natural life history dynamics or is a consequence of habitat alterations.

3.1.5 Hatchery vs. Wild Juvenile Life Histories in Shallow-Water Habitats

Marks and tags are used to designate hatchery fish from the progeny of wild spawners, with the caveat that hatchery marking is not uniform among regions. However, the proportion of marked hatchery fish has increased greatly during the last decade due to Congressional mandate, and the proportion of fin-clipped subyearling Chinook salmon released from hatcheries has risen from approximately 11–14% from 2002 to 2004 to 37.5% in 2005, 63% in 2006, and ~65% to 80% in 2007 and 2008, respectively (Bottom et al. 2011, collated from the Pacific States Marine Fish Commission Risk Management Information System database, http://www.rmpc.org/). This increase is reflected in field samples. Roegner et al. (2012) showed an increase in the proportion of fin-clipped subyearling Chinook salmon from a mean (± SD) of 8.0% ± 2.0% sampled from 2002 through 2006 to 53.2% ± 13.7% sampled in 2007 and 2008, with the variation occurring across sample sites. Marking of yearling fish, in contrast, remained relatively

high (71 to 74%) throughout the monitoring period. Similarly, the proportion of marked subyearling Chinook salmon at the Russian Island emergent-wetland increased from less than 1% from 2002 to 2005 to 23%, 51%, and 32%, respectively during the next 3 years (Bottom et al. 2011). Studies after 2009 likely have a more complete assessment of the hatchery component in Chinook salmon samples. Regardless of the increase of marked hatchery salmon, an unmarked fish still cannot be unequivocally designated the progeny of fish spawning in the wild because not all hatchery fish are marked.

The proportion of marked fish varies substantially among sample sites. In the Sandy River delta region (reach G), Sather et al. (2011) found very low capture percentages for marked fish: only 25% of all salmon and 8% of Chinook salmon captured via beach seines were marked. Likewise, in Cathlamet Bay wetlands and above (reaches B−D), only 148 of 5273 (2.8%) Chinook salmon collected in all wetland channels from 2002 to 2008 were marked (Bottom et al. 2011). Most of these fish were small and presumably progeny of wild origin (because hatcheries generally release fish >60 mm). In reach C, Jones et al. (2010) found a low percentage of hatchery marks at Lord/Walker Island (4.4%) and Ryan Island (4.4%), and moderate marking rates at White Island (18%). At Cottonwood Island (reach D), marked Chinook salmon composed <33% of the total salmon catch from April to December 2010 (Diefenderfer et al. 2011). In comparison, 85% of Chinook salmon were adipose-fin clipped at Deer Island (reach E), where there was only a small proportion of unmarked fry (Poirier et al. 2009a, b). Comparing wetlands formed after natural breach events in Cathlamet and Youngs Bays, G. Johnson et al. (2010) found 20% of Chinook salmon were unclipped fry and 58.9% of the total catch were hatchery marked. As noted above, Roegner et al. (2012) observed annual and also site-specific differences in marking rates at main-stem beach seine sites in reaches A and B from 2002 to 2008, but overall, up to 30% of Chinook salmon sampled from lower river and estuary sites were ≤60 mm and likely wild origin fish. As another line of evidence, stable isotope ratios from a subset of these fish indicated that most had consumed hatchery feed (Maier and Simenstad 2009). In contrast, Weitkamp et al. (2012) determined that 91 to 99% of Chinook and coho salmon and steelhead trout sampled from mid-water purse seines in reach A were of hatchery origin. Note these salmon were generally larger than those found in shallow water sites and most were actively outmigrating. Variations in the proportion of hatchery fish may reflect the proximity of study sites to hatcheries and natal streams. For example, in Grays River, few marked coho or Chinook salmon were present but hatchery-reared chum were discernible from wild fish by their size-at-date (Roegner et al. 2010). Almost all studies found the mean size of marked fish to be greater than that of concurrently sampled unmarked fish.

Overall, releases of more than 100 million summer, spring, and fall Chinook salmon produced by 72 artificial propagation programs throughout the Columbia River basin (HSRG 2009) are a major factor influencing contemporary patterns of estuary use by juvenile Chinook salmon. Artificial propagation programs and rearing practices to a large extent drive temporal and spatial patterns of salmon abundance, stock composition, and size distribution within the estuary (Bottom et al. 2011). By selecting for body size and time of estuary entry, hatcheries further influence salmon habitat use and residence times within the estuary (Campbell 2010; Bottom et al. 2011). Although RME programs have documented varying levels of overlap in the distributions of hatchery- and naturally-produced salmon (e.g., Johnson et al. 2010; Bottom et al. 2011; Diefenderfer et al. 2011), possible negative competitive or other ecological interactions between them have not been investigated on site- or estuary-wide scales. The potential influence of hatchery production programs on the success of estuary restoration programs remains unclear.

3.2 Habitat Opportunity Limitations

Habitat opportunity is defined as the availability of environments salmon can access and from which they can benefit (Simenstad and Cordell 2000; Gray et al. 2002; Bottom et al. 2005, 2011). Habitats include shallow-water main-stem Columbia River and tributary beaches, backwater sloughs, as well as emergent marsh, scrub-shrub, and forested swamp wetlands. Elements for evaluating habitat opportunity include physical constraints to connectivity (migration barriers, water depth), and physiological limitations set by water-quality parameters (primarily temperature and dissolved oxygen). Many restoration efforts focus on restoring habitat opportunity by implementing hydrological reconnections.

Many systems with potential salmon habitat are inaccessible to salmon because of hydrological barriers. The overall area of blocked and restorable habitat has been estimated to exceed 45000 hectares (http://www.estuarypartnership.org/tidally-impaired-lands). Diefenderfer et al. (2011) noted that assessing passage barrier state is the least cost-prohibitive means of estimating the state of habitat opportunity, yet such data are rarely collected or collated.

The physical opportunity for juvenile salmon to access shallow-water habitats varies in large part due to water level, and the physical forcing of water level varies longitudinally. In the lower river, tides dominate the hydrography, and water level and inundation vary at semidiurnal and synodic periods. Farther upriver, the influence of tides decreases and flood events, which are more stochastic in time and space, can determine periods of access to shallow wetlands. These two physical drivers have consequences for habitat opportunity. For the Kandoll Farm restoration site, G. Johnson et al. (2010) calculated the “realized habitat opportunity”, defined as the integration of physical opportunity and seasonal migration period for Chinook, chum, and coho salmon. Realized habitat opportunity was found to be a fraction of the total available time due to tidal variations, while species and interannual variations were due to varied migration periods. Bottom et al. (2011) used hydrodynamic modeling to examine potential habitat opportunity under various scenarios of inundation, water velocity, and temperature at Russian and Lord Islands (reaches B and D, respectively). Tidal range moderated habitat opportunity at Russian Island (the lower river site), but river flow limited opportunity to a higher extent at Lord Island (because of seasonal and annual variations in river flow). However, even in tidally dominated areas, flooding events occur. Material exchange was found to be maximal during winter floods (G. Johnson et al. 2010).

On a smaller scale, restoration projects that investigated fish community response to hydrological reconnections included tide-gate removals (G. Johnson et al. 2005, 2006, 2010; Roegner et al. 2010; Haskell and Tiffan 2011) and tide-gate replacements with “fish friendly” designs (G. Johnson et al. 2005, 2006; J. Johnson et al. 2010, 2011). Systems with impaired connectivity often have depauperate fish communities (Roegner et al. 2010), or they have a high incidence of non-indigenous species (Poitier et al. 2009; J. Johnson et al. 2008, 2011). Restoration projects in the LCRE consistently demonstrate that enhanced hydrological reconnections improve hydrographic conditions (increased wetted area, lower temperature), and changes in fish populations are among the initial biological responses to hydrological reconnections. Studies commonly show increased diversity of the post-restoration fish community, including increased access of salmonids to intertidal wetlands. The abundance and diet of Chinook salmon as well as hydrological metrics in emergent and forested wetlands formed after natural breach events in Cathlamet Bay and Youngs Bay were found to be similar to the surrounding areas, indicating an increase in system connectivity after breaching (G. Johnson et al. 2010). However, while tide-gate replacements (not breaches) usually increase hydrological connectivity, they often have reduced exchange

compared to breaches or non-modified sloughs (G. Johnson et al. 2006, 2009; J. Johnson et al. 2008, 2011; Haskell and Tiffan 2011). Greene et al. (2012) compared various types of tide gates on juvenile Chinook salmon density (primarily in Puget Sound but also in Youngs Bay), and found that all types of tide gates substantially limited salmon habitat use relative to non-gated reference areas. Increased connectivity also does not guarantee increased habitat use because the upstream source of potential migrants varies among tributary systems and main-stem reaches (G. Johnson et al. 2006, 2009). The proximity of a restoration or other habitat type to the migration corridor affects the genetic stock of origin, size structure, and prevalence of hatchery-raised fish observed (G. Johnson et al. 2006, 2009; Roegner et al. 2010), and the potential source of migrants in an important consideration for restoration site planning

3.2.1 Water Quality by Habitat Type

Dissolved oxygen (DO) concentration and temperature are the primary water-quality parameters thought to limit habitat opportunity and affect salmon performance.

3.2.1.1 Dissolved Oxygen

Relatively few studies have measured DO concentration in the LCRE, and the effects of low DO concentration on salmon were generally based on laboratory experiments, which are somewhat dated. These studies have found salmon to be relatively intolerant of low DO concentrations and to exhibit avoidance behaviors when exposed to oxygen levels of about 6.0 mg/L (Davis 1975). The limit to avoid acute mortality is 3.0 mg/L (EPA 2003). Whitmore et al. (1960) found juvenile Chinook and coho salmon avoided water with DO concentrations of 4.5 and 6.0 mg/L, respectively, and both species preferred DO concentrations of 9.0 mg/L. Sublethal effects include reduced growth and swimming speed. Growth reduction in Chinook and coho salmon occurred below about 7.0 mg/L and was severe for salmon exposed to DO levels below 4.0 mg/L. Low DO concentration progressively reduces swimming speed in juvenile salmon (by approximately 20% at 4.0 mg/L; Davis et al. 1963; Dahlberg et al. 1968), which can limit escape responses to predators.

Studies in the LCRE indicate DO concentrations are not limiting to salmon at most of the freshwater main-stem sites for which there are data (Table 3.1, Sather et al. 2009, 2011; Roegner et al. 2011a). However, DO concentration decreases below the 6-mg/L criterion in many poorly flushed sloughs and backwaters (e.g., J. Johnson et al. 2008; Poirier et al. 2009a, b; Sather et al. 2009, 2011; SBWC 2011). These occurrences of low DO concentrations are commonly associated with high temperatures in summer months, and they may limit salmon habitat use during the July through September period. In addition, and in contrast to conditions in the main-stem tidal freshwater habitats, ocean water with very low DO concentrations is commonly advected into reach A during the summer upwelling season, and may lead to increased stress and/or behavioral modifications in migrating juvenile salmon just before ocean entry (Roegner et al. 2011a). Thus, DO concentration levels in the Columbia estuary and backwater areas, while unlikely to be acutely lethal to salmonids, probably invoke behavioral and physiological responses that limit habitat use and/or reduce salmon performance and increase stress. More research is required to elucidate the behavioral and sublethal effects of low DO concentrations on salmon in the LCRE.

3.2.1.2 Temperature

Temperature is the paramount water-quality variable associated with delimiting acceptable salmon habitat (Bottom et al. 2005; Fresh et al. 2005). Review studies, which are typically based on laboratory

settings, suggest optimal temperatures for Chinook salmon range from about 10 to 16°C (McCullough 1999; EPA 2003; Richter and Kolmes 2005; Bottom et al. 2005). Temperatures below 10°C generally result in reduced growth rates, with lethal low temperatures ranging from 0 to 6°C, depending on acclimation temperature (Brett 1952; McCullough 1999). Temperatures in the range of 16 to 19°C may induce sublethal effects such as decreased growth and increased disease prevalence. Temperatures >19°C are generally considered potentially stressful. The incipient lethal temperature for juvenile salmon is between 23 and 25°C (Brett 1952; Baker et al. 1995). The critical temperature values and the categories presented here vary between the species, genetic stock, geographic origin, and life history type tested, as well as the experimental protocols employed such as acclimation period (McCullough 1999; EPA 2003; Richter and Kolmes 2005). To this point, establishing temperature tolerances and the effects of temperature stress on salmon physiology specifically for Columbia River salmon is a critical research need.

In the LCRE, temperature trends in the tidal freshwater portion of the LCRE are driven mainly by riverine processes, while temperature in the saline estuary is controlled by both riverine and oceanic inputs that are partially controlled by climate variability (Roegner et al. 2008, 2011b). As a consequence, there is a much wider daily variation of temperature (and salinity) in the estuary than in the tidal freshwater areas.

Seasonally, river water temperatures in the LCRE follow a regular and recurring pattern of low temperature (<10°C) during late November through early April. Optimal temperatures (10−16°C) occur from April through May, but they increase and remain >19°C from June through September or early October. Summer maximum temperature at Bonneville Dam can approach 24°C, near the reported incipient lethal temperature. By late autumn through winter, river temperature declines to about 6°C. Thus, seasonal temperatures regularly lie outside the thermal optima reported for salmon, and mean daily maximum temperatures approach the reported lethal levels. However, note that temperatures in shallow intertidal and subtidal areas and sloughs of restricted water exchange can diverge substantially from the main-stem river water due to atmospheric heating or cooling, groundwater flow, and biological effects such as shading by vegetation (e.g., Roegner et al. 2010; Bottom et al. 2011). This wide seasonal and local-scale range of temperatures has consequences for poikilothermic salmon, but few studies in the LCRE have directly investigated the effects of high temperature on salmon condition or physiology. Studies instead have correlated salmon abundance with temperature levels or modeled temperature to evaluate habitat opportunity.

A general theme found in both restoration sites and other surveyed systems is a decline in Chinook salmon abundance at intertidal and shallow-water habitats as temperatures reach about 19°C, which occurs each year around June. Bottom et al. (2011) modeled habitat opportunity based on temperature and depth criteria and found that high temperature limited habitat opportunity. Storch et al. (2011) noted that modeled growth of Chinook salmon was reduced in shallow-water habitats in the vicinity of the Sandy River delta during sustained periods of high temperature. At Deer Island (reach E) and Kandoll and Johnson wetlands (reach B), the 7-day mean maximum temperatures (7-DAM) exceeded 16°C by mid-late May, and were >19°C within the first week in June. Sagar et al. (2012) observed consistent declines in Chinook salmon density at sites in Reach C, E, F, and H beginning in June or July, when temperatures began to exceed 19°C. Stock-specific migrations indicate many salmon groups migrate in spring before temperatures reach stressful levels (Bottom et al. 2011). Likewise, chum migration is generally complete before temperatures reached 16°C (Roegner et al. 2010, 2012; Sagar et al. 2012). However, although abundances in intertidal wetlands fall to near zero in many sites after June, salmon are

still captured in subtidal areas and at main-stem shallow-water sites at temperatures as high as 22 to 24°C (Roegner et al. 2010, 2012; Bottom et al. 2011; Haskell and Tiffan 2011; Sagar et al. 2012). In reaches A and B, temperatures reached levels reported to be stressful to salmonids from late June through October of each year between 2003 and 2007 (Bottom et al. 2011; Roegner and Teel in review). During that period, 32.7% of the Chinook salmon were sampled from water >16°C, and 12.3% were sampled from water >19°C. Some evidence indicates salmon can benefit from increased temperature regimes. J. Johnson et al. (2009) found extremely high growth rates (1.29 to 1.62 mm/d) in waters of restricted exchange and high temperature. Roegner and Teel (in review) determined salmon caught during summer high-temperature periods had a high morphometric condition index, contrary to expectations of decreased fitness due to temperature-induced stress. High growth and condition may be maintained at high temperature with adequate oxygen and food supply (Myrick and Cech 2004), conditions apparently met at most monitored sites. More research is required to elucidate the effects of high temperature in the LCRE on salmon tolerance, behavior, and fitness, especially in relation to generic stock of origin.

3.3 Habitat Capacity Limitations

Habitat capacity is defined as the ability of a habitat to support functions benefiting salmon (Simenstad and Cornell 2000; Gray et al. 2002; Bottom et al. 2005). Positive factors defining habitat capacity include prey production and the resultant bioenergetic potential, while negative attributes include the presence and impacts of predators and competitors. Research to date has focused mainly on prey resources, and several other studies have investigated competitive interactions. Predatory interactions in wetlands have not been specifically studied.

3.3.1 Prey Availability and Bioenergetic Potential

Most diet studies in the LCRE have concluded that insects (particularly chironomid flies, see below) are the main prey for juvenile salmon, especially for smaller salmon <80 mm that inhabit shallow-water systems. Insect production in wetland habitats is seasonally variable (highest in late summer) and is generally found to be substantial, although the highest production occurs when salmon use of wetlands is at present reduced. At Russian Island, emergence rates of total insects tended to increase with time (max 140 ind m^{-2} d^{-1}), while chironomids peaked in June (~40 ind $m^{-2}d^{-1}$) (Ramirez 2008). Across 12 wetland sites, mean annual insect density estimates from fall-out traps ranged from 551 to 4365 ind/m^2 (Bottom et al. 2011). Haskell and Tiffan (2011) found insect production was enhanced and overall invertebrate diversity was increased post-restoration at Crims Island. Insects including chironomids have also been shown to be exported from wetland production sites to the larger ecosystem (Ramirez 2008; Eaton 2010; G. Johnson et al. 2010), further emphasizing the important trophic link between wetland prey production and migrating salmon. A bioenergetic modeling approach by Storch (2011) focusing near the Sandy River delta area suggests temperature rather than food supply limits juvenile salmon growth. These studies indicate prey production is high and not limiting in individual wetland habitats. However, the overall loss of marshes in the LCRE and the reduction of a macrodetritus-based food web may have reduced the overall capacity of the system to support juvenile salmon compared to historical levels (Maier and Simenstad 2009).

3.3.2 Predators

Shallow-water habitats are often considered refuges from predation, but few studies located for this review specifically investigated predation by fish or birds on juvenile salmon in shallow-water areas of the LCRE. Piscine predators of sufficient size to consume juvenile salmon are rarely identified in species lists (e.g., Bottom et al. 2011; Jones et al. 2010; Sather et al. 2011), which may be an artifact of beach seine sampling. Haskell and Tiffan (2011) concluded that restoration at Crims Island did not greatly benefit northern pikeminnow (*Ptychocheilus oregonensis*), bass (*Micropterus* spp.), or walleye (*Sander vitreus*). Conversely, other records do indicate the presence of predators. Pikeminnow made up 6%, smallmouth bass 1.0%, and largemouth bass ~0.05% of the total catch near the Sandy River delta; some individuals were large enough to consume salmon (Sather et al. 2011). Similarly, at Franz Lake in Reach H, pikeminnow made up 6.3% and smallmouth bass 1.2% of the total catch, but these species were less abundant at other sites in Reach H or in sampling sites at Reaches C, E, and F (Sagar et al. 2012). In passive integrated transponder (PIT)-tag arrays at Cottonwood Island (rkm 113), Diefenderfer et al. (2011) detected the presence of tagged northern pikeminnow at the mouths of wetland channels (sizes not reported) mainly during May and June, concurrent with high salmonid catches. Roegner et al. (2010) found yearling coho consumed chum fry in restoring wetlands in Greys River. Weikamp et al. (2012) note that with the exception of cutthroat trout and adult Chinook salmon and steelhead, most fish species caught in purse seines in deeper areas of reach A were too small to feed on migrating juvenile salmonids. Other potential predators on juvenile salmon include California (*Zalophus californianus*) and Stellar (*Eumetopias jubatus*) sea lions, and harbor seals (*Phoca vitulina*), but their impact has not been documented outside of areas below Bonneville Dam.

However, predation on juvenile salmon by birds, especially double-crested cormorants and Caspian terns roosting in the lower estuary, has been significant. Recent studies estimated millions of steelhead and yearling and subyearling Chinook salmon were eaten annually by birds roosting on a single nesting island (Collis et al. 2001; Ryan et al. 2003; Roby et al. 2003; Anderson et al. 2007). Many larger yearling and smolted subyearling fish are presumably taken in main channel habitats (Harnish et al. 2012). Four to 8% of fish tagged in the Kalama River bypassed PIT-tag receivers on Cottonwood Island but were detected on East Sand Island bird colonies (Diefenderfer et al. 2011). Bird predation of “tule” stock subyearling Chinook salmon is especially significant. Sebring et al. (2010) found up to 44% of salmon released from lower river hatcheries in summer were taken by birds. Harnish et al. (2012) found bird predation was estimated to account for 5.0, 5.5, and 17% of subyearling and yearling Chinook salmon and steelhead, respectively, in experimental releases from Bonneville Dam. This predation is (probably) the largest and (certainly) the best-documented source of mortality of juvenile salmon in the LCRE, but it is unknown to what extent this predation is occurring in wetland habitats.

3.3.3 Competitors

Competitive interactions between salmon and other fish species have been studied in Cathlamet Bay marshes, Grays River habitats, and the Sandy River delta region. By far the most abundant fish species in shallow-water LCRE habitats is threespine stickleback (*Gasterosteus aculeatus*), which generally composes >90% of the total fish catch in most studies. Spilseth and Simenstad (2011) compared competitive overlap between salmon and stickleback within Russian, Wallace, and Lord islands, and, based on consumption rates and available prey resources, overlap was considered limited. Eaton (2010) examined resource partitioning between Chinook, chum, and coho salmon at intertidal and subtidal

habitats of the Grays River system. Chum had high spatial and dietary but low temporal overlap; Chinook and coho salmon exhibited spatial segregation. Sather et al. (2012) investigated diet overlap near the Sandy River delta (rkm 190–208) between juvenile Chinook salmon and resident species including threespine stickleback, banded killifish (*Fundulus diaphanous*), bluegill (*Lepomis macrochirus*), and pumpkinseed (*Lepomis gibbosus*). Results suggest little overlap in the diets of juvenile Chinook salmon and the diets of the resident species examined. Data to date suggest competitive interactions may be of limited consequence to salmon. However, these conclusions are based on few studies and could reflect incomplete research.

3.4 Salmon Performance

The performance of salmon in a wetland is a synergism of opportunity and capacity indicating accrued benefit (Simenstad and Cordell 2000; Gray et al. 2002; Bottom et al. 2005). Metrics defining performance are measures of fish enhancement, such as diet and foraging success, residency and growth, condition, and life history diversity.

3.4.1 Diet and Foraging Success

While juvenile salmon are omnivorous feeders that prey on a wide variety benthic, epibenthic, planktonic, and neustonic organisms, the many diet studies in the LCRE convincingly demonstrate that insects, primarily chironomids, are the single most important prey type. Lott (2004) and Bottom et al. (2011) found chironomids dominated diet compositions in wetlands and were the most important prey taxa. Gammarid amphipods are also prevalent prey, especially in main-stem and subtidal areas (Maier and Simenstad 2009: Bottom et al. 2011). At sites in the Sandy River delta (rkm 190–208), Storch and Sather (2011) found dipterans, hemipterans, amphipods, and mysids generally had the highest relative importance (%IRI) in fish diets. Both insects and amphipods are energy-dense and are likely excellent food for salmon (Gray 2005; Storch 2011). Maier and Simenstad (2009) used stable isotopes to investigate salmon food webs, and (aside from the high contribution from hatchery feed) they found vascular plants composed a primary source of material for the food chain (transmitted to salmon by their insect and amphipod prey). Juvenile salmon tend to feed heavily in wetland areas; stomach contents of salmon sampled at wetland (as well as main-stem) sites were generally between 70 and 95% full (Bottom et al. 2011). However, smaller subyearling Chinook salmon primarily used marsh plains and intertidal channels, whereas larger subyearlings occupied deeper subtidal channels (Bottom et al. 2011; Haskell and Tiffan 2011), and dietary changes in summer have been attributed to salmon leaving shallow wetlands feeding on more pelagic prey, notably cladocerans (Anderson 2006; Haskell and Tiffan 2011). In the brackish estuary, food of marine origin becomes common in diets (Roegner et al. 2008; Maier and Simenstad 2009; Bottom et al. 2011), indicating that subyearling salmon feed throughout their seaward migration. Restoration projects can also broaden salmon diets. Subyearling Chinook and coho salmon and yearling coho all tended to have more diverse diets in newly restoring restoration sites than in the surrounding channel habitat (Roegner et al. 2010).

3.4.2 Migration, Residency, and Growth

Outside of specific experiments on field-captured salmon, most tagged fish in the LCRE are hatchery-reared (i.e., large yearling or smolted subyearlings), and tagging studies were designed to measure migration timing and estimate survival through various hydropower systems. Results derived from these

studies suggest habitat use in the LCRE is dependent on stock, size, and degree of smoltification. Large smolts migrate rapidly through the system and can transit from Bonneville Dam to the estuary in <5 d (Dawley et al. 1986; Magie et al. 2008; Harnish et al. 2012). Small sized individuals and especially local stocks migrate more slowly and use shallow-water habitats for rearing. Here we summarize tagging studies from main-stem and shallow-water environments.

3.4.2.1 Performance in Main-Stem Channel Habitats

Starting with the work of Dawley et al. (1986), most studies used marks or tags (acoustic, radio, or PIT) to track larger subyearling or yearling salmon that often migrate rapidly through the LCRE. In the lower Willamette River, Friesen et al. (2007) found yearling Chinook salmon moved with median travel times of 11.3 km/d. Of 981 acoustically tagged juvenile salmonids in spring and summer, only 7% used side channels within the Sandy River delta, and most tagged fish migrated rapidly in the main-stem channel (Sather et al. 2009). Residence time was longer for subyearling fall Chinook salmon (1−10 h) than for yearlings (0.5−2 h), and steelhead had the shortest residence times (<0.5 h). McMichael et al. (2011) and Harnish et al. (2012) used acoustic telemetry to track subyearling and yearling Chinook salmon and steelhead from Bonneville Dam to the ocean. Survival for release groups was relatively high from rkm 238 to rkm 86 and decreased thereafter. Overall survival was 0.64, 0.78, and 0.53 for subyearling and yearling Chinook salmon and steelhead, respectively, with subyearling survival decreasing over time. Mean travel times for the salmonid groups were rapid (3.1 to 4.1 d), but fish often decreased their migration rate near the Astoria Bridge (rkm 22). Similarly, at Jones Beach (reach C), Magie et al. (2008) used a paired trawl and PIT-tag detector to investigate survival and transit time of salmonids from Bonneville Dam. At Jones Beach, they detected 3.3% of fish recorded at the dam; median travel time across years was approximately 4 d. In contrast, at shallow-water main-stem sites, Roegner et al. (2012) caught only three PIT-tagged salmon in 6 years of sampling (out of ~12,000 salmon captured), indicating the larger and smolted fish that are commonly tagged generally are less commonly found in shallow-water habitats (e.g., Weitkamp et al. 2012).

3.4.2.2 Performance in Shallow-Water Habitats

The residence time of juvenile salmon in shallow-water habitats appears to be dependent on the location and timing of the study as well as the origin and life history strategies of the population examined. For example, during spring and summer 2007 and 2008 the residence time of acoustically tagged steelhead and yearling and subyearling Chinook salmon (>95 mm) at the Sandy River delta (rkm 190−208) occurred over a matter of hours. These fish were tagged at upstream locations above Bonneville Dam (Sather et al. 2009). In contrast, during winter and early spring months 2010 and 2011, the mean residence time of juvenile Chinook and coho salmon in the same area ranged from approximately 24−34 d, with a median residence time ranging from 11−26 d for the different tagged groups (G. Johnson et al. 2011; Sather et al. 2012). Fish tagged for the winter-early spring residence time study were captured directly at the site of release, suggesting site fidelity (G. Johnson et al. 2011; Sather et al. 2012).

Both coded-wire tags (CWTs) and PIT tags have been used to elucidate migration and residency. CWT recaptures at beach seine sites in reaches A and B revealed maximum migration times of 143 d for subyearling and 52 d for yearling fish (Roegner et al. 2012). PIT tags have been used experimentally to measure residence in natural, restored, and degraded systems. At intertidal channels at Russian Island,

PIT-tagged or batch-marked Chinook salmon had average residences of 5 to 7 d with maximum residence times of 26 to 34 d (Bottom et al. 2011). Of the recaptured salmon in 2006 and in 2008, 37% resided for at least 1 week, 14% resided for at least 2 weeks, and 5% resided for at least 3 weeks in the intertidal channels. (Note that these residency measurements indicate local and not site-specific uses, because salmon must leave when the site is dewatered). Similarly, experimental releases of PIT-tagged hatchery Chinook salmon above the tide-gated slough of Tenasillahe Island revealed a wide range of residence times (1−119 d) with a median between 41 to 45 d for the various release groups (J. Johnson et al. 2008). In contrast to these studies, at shallow backwaters sites at Crims Island, Haskell and Tiffan (2011) found juvenile salmon used areas for one or two tidal cycles, and Diefenderfer et al. (2011) found the average residence time of fish detected around Cottonwood Island was 8.9 hours (SD = 26.1 hours).

Decoding tags indicates a wide diversity of stocks use shallow-water environments. While PIT-tag receivers at Deer Island recorded few detections, indicating little out-of-watershed use, Diefenderfer et al. (2011a) found the geographic origins of the salmon were diverse in shallow-water habitats around Cottonwood Island. Based on the presence of hatchery marks, Roegner et al. (2010) concluded out- of-basin Chinook salmon were using restoring wetlands in Grays River. Both CWT and genetic data confirmed fall Chinook salmon from coastal estuaries occasionally entered into reach A (Roegner et al. 2012).

Another method of assessing residency in saline water uses strontium otolith elemental analysis that indicates contact with saltwater. From Chinook salmon collected year-round between 2003 and 2005 in reach A, Campbell (2010) and Bottom et al. (2011) estimated residency in saline water ranged from 0 to 176 d, and sizes at the estuary entrance were estimated to be between 34 and 178 mm. Fifty percent of salmon in 2004 and 2005 entered saltwater as fry (≤60 mm) and residence times decreased as the size of juvenile Chinook salmon increased. As a comparison, Miller (2011) determined from otolith microchemistry that adult Chinook salmon returning to the Sacramento River in California were derived from subyearling migrants that entered saltwater over a range of sizes: 20% were <55 mm, while 48% left freshwater when between 55 and 75 mm, and the remainder left when >75 mm.

Direct measurement of salmon growth rates requires recapture of tagged individuals, which few studies in the LCRE have performed during the review period. Friesen et al. (2007) found both yearling and subyearling migrants increased in size from upstream to downstream sampling sites in the Willamette river, suggesting growth during migration. In the Russian Island environment, Chinook salmon had growth rates of 0.60 and 0.67 mm/d, and individuals that resided more than 2 weeks increased in fork length (FL) by an average of 13.8 mm (Bottom et al. 2011). J. Johnson et al. (2009) monitored PIT-tagged hatchery Chinook salmon in a low connectivity slough at Tenasillahe Island with marginal water quality and found an average residency of 42 d and extremely high growth rates (1.29 to 1.62 mm/d).

Estimates of growth are also made by analyzing otolith or scale increments. Based on otolith widths, Campbell (2010) found growth rates of juvenile Chinook salmon ranged from 0.35 to 0.49 mm/d. Claiborne et al. (2011) evaluated the scales of adult Willamette Spring Chinook released from net pens in the estuary to investigate size-dependent mortality in the ocean. Larger fish (>150 mm) at ocean entry were found to return at a higher proportion than smaller fish during 2002 to 2004, but not in 2005. Larger fish also returned earlier than smaller growing fish. Sagar et al. (2012) found growth rates of juvenile Chinook salmon at tidal freshwater sites ranged from 0.47 to 0.61 mm/d, with growth rates in the lower range more commonly observed in fish collected in Reach C. Modeling can also lend insight into salmon growth potential. Storch (2011) used bioenergetics modeling to describe the growth of juvenile Chinook

salmon near the Sandy River delta (rkm 190−208). Modeled outputs indicate the growth of fish in these habitats was generally positive, except during time periods that coincided with temperature extremes (e.g., winter and summer months).

Finally, the condition of salmon has been determined using both morphological indices and biochemical metrics. At Crims Island, subyearling Chinook salmon were larger and had higher condition factors (Fultons's K, W/L^3) at the restoration site compared with pre-restoration conditions (Haskell and Tiffan 2011). Roegner and Teel (in review) used residual analysis to evaluate relative fitness and found a lower condition index of salmon sampled in the estuary than those sampled contemporaneously in tidal freshwater reaches. L. Johnson et al. (2007) found both the lipid content and condition factor of yearling and subyearling Chinook salmon decreased from upstream tidal freshwater sites to reach A; some fish contained <1% lipid per body weight (a level associated with increased mortality) (Biro et al. 2004). Sagar et al. (2012) examined lipid content in both marked and unmarked juvenile Chinook salmon and observed that the decline in lipid content from upstream to downstream sites was most pronounced in marked fish, and was less consistently observed in unmarked fish. Lipid content and morphological condition also tended to be low in juveniles captured early in the sampling season (e.g., in April as compared to those collected later, (e.g., in May and June), consistent with results from Roegner and Teel (in review). It has yet to be determined if these variations in salmon condition relate to subsequent survival.

3.4.3 Contaminants and Salmon Health

Concentrations of organic contaminants, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethanes (DDTs) and polybrominated diphenyl ethers (PBDEs) are generally low and non-threatening in hatchery feed and within hatchery juvenile Chinook salmon (L. Johnson et al. 2010); however, substantial proportions of specimens caught in the lower river have exposure levels to one of more of these contaminants exceeding values thought to cause health risks (LCREP 2007; L. Johnson et al. 2007, 2013; Sloan et al. 2010; Yanagida et al. 2012). For example, concentrations of PAH metabolites were above estimated effect thresholds (Meador et al. 2008) in over 40% of juvenile Chinook salmon bile samples from the lower Columbia River (Yanagida et al. 2012). Moreover, ~50% of subyearling fall Chinook samples from tidal freshwater sites (Johnson et al. 2013) and ~66% of Chinook smolts from the saltwater portion of the estuary (Johnson et al. 2007) had PCB concentrations exceeding the 2400 ng/g lipid threshold estimated by Meador et al. (2002). Maximum concentrations of PCBs, DDTs, and PBDEs in juvenile salmon from the lower Columbia were all within the upper range of juvenile salmon sampled in the Pacific Northwest, and the condition and lipid content of a number of these fish, especially smolts, was also reduced. Body lipid content can influence the tolerance of an organisms to bioaccumulative contaminants, and individuals with lower lipid content typically show a greater toxic response to comparable exposure (Lassiter and Hallam 1990). Consequently, L. Johnson et al. (2007, 2013) and Arkoosh et al. (2010) suspect the decline in lipid content described above could increase the sensitivity of fish to the effects of these bioaccumulative contaminants. The health of juvenile salmon may also be affected by exposure to other classes of contaminants present in the lower Columbia River, including pharmaceuticals and personal care products in wastewater (LCREP 2007; Morace et al. 2012); current use pesticides (NMFS 2008) and toxic metals such as copper (Hecht et al. 2007). More work on the effects of contaminants on salmon health is warranted, and especially to ascertain whether restoration projects near contaminated sites will benefit or harm migrating fish.

3.5 Conclusions

Based on evidence to date, the primary direct beneficiaries of restoration of main-stem wetland habitats will be small subyearling Chinook and chum salmon with smaller numbers of larger yearling Chinook salmon found in shallow areas. Juvenile Coho salmon are more prevalent in tidal wetlands within tributary systems than in main-stem sites. Many of the small juvenile salmon are wild spawned, and constitute a life history type not represented by the hatchery production system. Restoration of main-stem wetland habitats also has indirect benefits to juvenile salmon through export of organic materials, nutrients, and prey resources from shallow-water to main-tem areas. In order to restore life history diversity to Columbia River salmon populations, it is critical to protect, restore, and enhance the wetland habitat upon which these fish depend.

Habitat opportunity appears to be a major limitation to salmon performance. Many potential systems are simply unavailable due to migration barriers. Reduced flushing, leading to high-temperature and low-oxygen conditions, also appears to limit the time salmon can benefit from wetland habitats during summer months. Tide gates, even those with "fish friendly" designs, improve access but are not as beneficial as more open hydraulic reconnections for either salmon movements or for maintenance of adequate water-quality parameters. Conversely, habitat capacity and the limited information about salmon performance in wetland sites indicate salmon are benefitting from wetland food production that results in relatively high growth rates. Wetland-derived insect prey also appears to be regularly transported to the wider ecosystem, where it is available to fish not inhabiting wetlands. However the overall loss of marshes in the LCRE and the reduction of a macrodetritus-based food web may have reduced the overall capacity of the system compared to historical capacities. Competition and predation within wetlands requires more research but present data have not documented adverse effects on salmon performance. Additional research is needed, including potential direct or indirect interactions with non-native species. Predation studies have not been conducted in wetland sites, and bird predation in particular may be significant. Nonetheless, restoration activities that increase habitat opportunity are likely to benefit many salmon populations, and effort should be directed toward targeting sites that can be fully reconnected rather than left with restricted hydraulic connections.

Patterns of estuary habitat use and the life histories of juvenile salmon are directly tied to their freshwater sources. Large releases of salmon from hatchery sources are a major driver of contemporary stock abundances and the arrival times, sizes, habitat preferences, and residence times of juveniles in the estuary. Because hatcheries target relatively few salmon stocks and phenotypes, the dominant estuary rearing behaviors today may or may not reflect the habitat and restoration needs of under-represented and at-risk stocks. Furthermore, neither the interactions of hatchery- and natural-origin salmon nor the potential effects of hatchery releases on the estuary ecosystem have been investigated. It is unclear, for example, whether continued subsidies of similarly-sized hatchery smolts released in concentrated pulses during the spring have enhanced bird or other predator populations in the LCRE.

4.0 Factors in the Estuary that Limit Recovery of At-Risk Salmon Populations and ESUs

The long-term viability of salmon populations has been defined based on four performance criteria: abundance, productivity, spatial structure, and diversity (McElhaney et al. 2000). The estuary contributes directly to each of these variables (Bottom et al. 2005; Fresh et al. 2005). The amount of estuarine habitat that is available to juvenile salmon influences population abundance and productivity. The distribution, connectivity, and variety of habitat features in the estuary contribute to the diversity and the spatial structure of each salmon population. As juveniles migrate through the estuary and to the ocean, the quality of estuarine habitats can influence the diversity and productivity of populations through life-stage specific survivals (Fresh et al. 2005).

Accounting for the estuary's influence on population viability requires methods for reconstructing the estuarine life history pathways and performance of individuals from particular populations or ESUs of interest. Until recently, the movement patterns and upriver sources of fish captured in the estuary could be determined only for tagged individuals (e.g., using CWTs, PIT tags, or acoustic tags). Yet most tagging methods are limited to large (e.g., >95mm) subyearling and yearling size classes, whose estuarine life histories and performance appear to differ markedly from those of smaller subyearlings (Bottom et al. 2011). Representative tagging of naturally produced salmon from target populations is quite difficult, and recapture rates in the estuary are generally low due to limited sampling, thus hampering the ability to draw inferences about estuary performance using mark/recapture methods alone.

Within the last decade, stock identification techniques using microsatellites as genetic markers have improved sufficiently to classify the genetic sources of Chinook salmon (i.e., tagged or untagged) captured in the estuary (e.g., Teel et al. 2009). The techniques are allowing investigators to compare estuarine life histories and ecology among genetic stock groups and ultimately, may allow the development of restoration strategies targeting the particular habitat needs of at-risk stocks and populations. However, several challenges still limit interpretation of the stock affiliations of individual fish based on genetic composition. First, the existing genetic baseline can distinguish genetic sources to approximately the ESU level, but it cannot resolve differences at finer geographic scales (e.g., individual streams of origin). Improved baselines for individual Columbia River subbasins could increase the geographic resolution of genetic analyses in the future. Second, artificial propagation and fish stocking programs have redistributed many stock groups outside their natal basins (e.g., Teel et al. 2009; Johnson et al. 2011; Sather et al. 2011; Bottom et al. 2011; Roegner et al. 2012) and will continue to complicate interpretations of the geographic origins of individuals sampled in the estuary. For example, Johnson et al. (2011) described past stock transfers of "tule" fall Chinook from the Big White Salmon River into hatchery and tributary populations in the Columbia River Gorge and lower river. Similarly, upper Columbia River Summer/Fall stock (from the upper river east of the Cascades) are now produced in Columbia River Gorge tributaries and hatcheries and in main-stem areas below Bonneville Dam. These and many other fish transfers demand caution before drawing conclusions about the geographic origins of estuary-resident juveniles based solely on their genetic composition.

New analytical methods have successfully reconstructed the sources of fish from areas with sufficient environmental heterogeneity to leave distinct chemical signatures on otoliths. For example, variations in otolith strontium isotopic ratios ($^{87}Sr/^{86}Sr$) were found among major California Central Valley rivers and hatcheries, enabling classification of fall-run ESU Chinook salmon to their natal stream sources (Barnett-

Johnson et al. 2008). Ultimately, a combination of genetic and otolith techniques may improve interpretations of the geographic origins of salmon. Barnett-Johnson et al. (2010) combined broad-scale genetic and fine-scale otolith isotopic markers to distinguish the geographic origins of Chinook salmon from the Mid and Upper Columbia River summer/fall run. Other otolith isotopic or elemental signatures may enable finer-scale interpretations of natal stream origins for salmon from other Columbia River ESUs, but additional research is needed to develop these tools before they can be applied.

Difficulty distinguishing hatchery from naturally produced salmon remains a fundamental impediment for interpreting the sources of salmon collected in the estuary. The inability to fully account for hatchery subsidies in naturally spawning populations also may further mask declining population trends or yield false conclusions about the recovery of wild populations (R. Johnson et al. 2012). Until recently, the proportion of fish marked in Columbia River hatcheries was low and precluded comparisons between hatchery and naturally produced stocks within the estuary. For example, excluding releases of interior spring Chinook salmon stocks, hatchery marking rates for Columbia River subyearling Chinook salmon were only ~11 to 14% from 2002 to 2004. This increased significantly after 2006 to ~65% and 80% in 2007 and 2008, respectively (Bottom et al. 2011; www.rmpc.org). Despite the overall improvement, marking rates for some stocks remain relatively low, as mentioned in the previous section, limiting abilities to distinguish wild from hatchery salmon within the estuary. For example, <70% of the subyearling Upper Columbia River summer/fall Chinook salmon released from hatcheries were marked in 2008 and 2009 (www.rmpc.org). New analytical tools using otolith structural (Andrew Claiborne, Oregon State University, personal communication) and isotopic methods (R. Johnson et al. 2012) hold promise for distinguishing unmarked hatchery from wild fish, but these methods are costly, involve lethal sampling, and may require further validation for particular locales or stocks of interest.

Reconstruction of the life history pathways and the estuary performance of salmon from particular populations or stock groups is necessary to answer the question, “do factors in the estuary limit recovery of at-risk salmon populations and ESUs?” Recent improvements in genetic stock identification techniques have enabled researchers to identify and compare the estuarine distributions and habitat associations of juvenile Chinook salmon from a diversity of genetic stock groups. However, it is much more difficult to assess how habitat conditions and salmon performance within the estuary may affect adult returns or the viability of a particular population or stock group. Here we review the status of recent RME efforts to address each category of information need: 1) stock-specific habitat use and performance within the estuary (i.e., which estuarine habitats support each salmon ESU?), and 2) estuary contributions and limits to the viability of populations and ESUs (i.e., does salmon performance within the estuary limit the number of adults returning to particular ESUs?).

4.1 Stock-Specific Habitat Use and Performance Within the Estuary

A key objective of several estuary RME plans has been to characterize the stocks of origin for juvenile Chinook salmon occupying selected estuary habitats and regions. RME surveys primarily have targeted shallow-water habitats, including sandy beaches along the estuary’s main-stem; tributary confluences and deltas; and wetlands, tidal floodplains, and other backwater areas. An expanding genetic baseline for Chinook salmon and improved stock identification techniques have enabled estimation of the most probable stock affiliations of individuals in each sample collection. The reports of recent RME surveys thus provide the first snapshots of the proportional stock composition for Chinook salmon found at each estuary locale. While the methods for genetic stock identification have been standardized, the

sampling design and time periods of recent surveys have not, thereby limiting efforts to synthesize results across studies. Nonetheless, between-survey comparisons are useful for developing hypotheses about stock distributions and migrations at broader (i.e., interannual or estuary-wide) scales.

Improvements in the genetic baseline for Chinook salmon in the last decade have increased the regional specificity and accuracy of genetic assignments. Using a standardized West Coast Chinook salmon genetic baseline based on microsatellite DNA loci (Seeb et al. 2007) and additional data from previous Columbia River studies, nine regional Chinook salmon stock groups have been defined within the Columbia River basin. Genetics data for selected rivers outside the basin also have been incorporated into the genetic baseline to assess potential estuary contributions from Oregon and Washington coastal fall Chinook stocks and from hatchery releases or naturalized populations of Rogue River fall Chinook salmon (Bottom et al. 2011; Sather et al. 2011; Roegner et al. 2012).

4.1.1 Genetic Stock-Group Distributions and Habitat Associations

Comprehensive reports of Chinook salmon stock affiliations include summaries of 2002 to 2006 surveys of the lower estuary (rkm 8−101, reaches A−C) (Bottom et al. 2011; Roegner et al. 2012); 2009 to 2010 collections in the Cowlitz to Lewis River region (rkm 110−141, reaches D and E) (Sather et al. 2011); and 2007 to 2010 samples in the Sandy River delta (rkm 188−202, reach G) (Sather et al. 2009, 2011). The genetic results for each survey expressed as the proportional stock contributions for all stations and time periods are summarized in Table 4.1.

Table 4.1. Estimated Stock Composition of Chinook Salmon Reported From Surveys in the Lower Columbia River Estuary (rkm 8–84, reaches A−B) (Roegner et al. 2012), Mid-Columbia River Estuary (rkm 110–141, reaches D and E) (Sather et al. 2011), and Sandy River Delta (rkm 188–202, reach G) (Sather et al. 2011).

	Estimated Stock Proportions (%)				
	Lower Estuary(a) Jan 2002–Dec 2006	Middle Estuary Jan 2009–Feb 2010		Upper Estuary Jun 2007–Apr 2010	
Stock Group	(n = 2138)	Unmarked (n = 362)	Marked (n = 54)	Unmarked (n = 1242)	Marked (n = 159)
West Cascade Tributary fall	50.8	75.4	24.4	15.3	4.3
West Cascade Tributary spring	2.8	4.8	5.0	2.1	0.6
Willamette River spring	1.3	4.4	13.6	7.7	1.9
Spring Creek group fall	33.8	11.5	57.1	34.9	68.7
Deschutes River fall	0.3	0.3	0.0	3.2	1.8
Mid/upper Columbia River spring	0.0	0.0	0.0	0.0	0.6
Upper Columbia River summer/fall	6.2	0.2	0.0	33.4	19.6
Snake River fall	0.8	0.9	0.0	3.3	2.4
Snake River spring	0.1	0.0	0.0	0.1	0.0
Rogue River fall	2.5	0.3	0.0	0.0	0.0
Coast fall/spring	1.4	--	--	--	--

(a) Results are for beach seine sites surveyed in the lower estuary, 2002−2006. Additional data for wetland habitats are summarized by Bottom et al. (2011).

The genetic composition of more than 2100 Chinook salmon collected in the lower 100 km of the estuary was dominated by fall-run Chinook salmon from the lower Columbia River ESU; approximately 85% of all samples were classified as West Cascade Tributary or Spring Creek group fall Chinook stocks (Table 4.1; Bottom et al. 2011). However, each of the other stock groups in the genetic baseline also contributed to lower-estuary samples except for Middle and Upper Columbia spring-run stocks.

Sather et al. (2011) collected >400 genetic samples between January 2009 and February 2010 in mid-estuary reaches between the Cowlitz and Lewis river tributary junctions. The stock composition results for unmarked (n = 362) and marked hatchery Chinook (n = 58) were summarized separately. Although survey years and sample sizes differed for the two studies, the stock composition results for the lower and mid-estuary were similar (Table 4.1). For example, more than 80% of the unmarked and marked Chinook salmon collected in mid-estuary habitats were also classified as fall-run Chinook from the lower Columbia River ESU. Unmarked Chinook in the mid-estuary region were dominated by West Cascade Tributary fall run Chinook, but Spring Creek group fall Chinook represented a higher proportion of the marked salmon.

In contrast to the lower and mid-estuary results, West Cascade Tributary fall Chinook salmon composed a much smaller proportion (~15%) of recent sample collections (2007−2010, n=1401) from shallow habitats near the Sandy River delta (Table 4.1; Sather et al. 2011). When contributions from the Spring Creek group fall stock are included, the lower Columbia River ESU still accounted for ~50% of the marked and >70% of the unmarked Chinook salmon collected in shallow habitats of the Sandy River delta. However, unmarked salmon in the Sandy River delta vicinity included contributions from a greater diversity of genetic stock groups that were less prevalent in the lower and mid-estuary survey areas, including upper Columbia River summer/fall (~33%), Willamette River spring (~7%), Snake River fall (~3%), and Deschutes River fall (~3%) Chinook stocks. The presence of Willamette River spring Chinook in the Sandy River collections (upstream from the Willamette confluence) are likely a legacy of past hatchery releases of Willamette River spring Chinook in the Sandy River basin (Myers et al. 2006; Sather et al. 2011).

Teel et al. (2009) analyzed winter and spring sample collections from a study of main-stem habitats and floodplain wetland sites in the lower Willamette River just above the Columbia River confluence (Baker 2008). The results indicated that subyearling Chinook salmon from a diversity of Columbia River stocks move into the lower Willamette River to occupy shallow habitats. Not surprisingly, natal Willamette River spring Chinook salmon accounted for a large proportion (40 to 71%) of the winter and spring collections from river and wetland sites in 2005. However, subyearling spring and fall Chinook salmon from other ESUs also moved into the Willamette River from the Columbia River during winter and spring, including significant contributions from Spring Creek group fall Chinook (e.g., 49% of river-wetland samples in winter 2005), West Cascade Tributary spring and fall stocks, and the upper Columbia River summer/fall group (26% of the 2006 wetland sample).

The genetic composition of Chinook salmon smolts in the Grays River (reach B) included a mixture of Rogue River (63%), West Cascade Tributary fall (25%) and spring (8%), and Willamette plus Upper Columbia (2%) stocks, demonstrating the effects of past hatchery practices and straying on salmon populations (Roegner et al. 2010).

Genetic results for a variety of time periods and estuary regions since 2002 suggest that stocks of Columbia River Chinook salmon are not distributed uniformly in space or time, but exhibit characteristic

patterns of migration and habitat use. Samples from lower (Bottom et al. 2011; Roegner et al. 2012) and mid-estuary reaches (Sather et al. 2011) included contributions from the full diversity of stock groups (except for interior spring Chinook stocks), but were dominated by lower Columbia River fall Chinook salmon. In contrast, samples upriver near the confluence of the Sandy river delta included a greater representation of interior Columbia River and Willamette River stocks. Spring Creek group fall Chinook stocks were abundant primarily during the spring in both the Sandy River delta and the lower estuary surveys, but were nearly absent during the summer and fall. West Cascade Tributary fall stocks were well represented in both survey areas throughout the year, but were more dominant in summer and fall samples from the lower-estuary sites. The Upper Columbia River summer/fall stock group was present at the Sandy River delta during much of the year, but increased substantially in summer and fall collections, and accounted for 73% of the July samples. Although much less prevalent in lower-estuary surveys, the proportions of Upper Columbia River fall Chinook similarly increased during the summer-fall.

At a site scale, genetic survey results were generally more variable, and no consistent differences in stock proportions were apparent among shallow-water habitat types sampled within the same estuary regions. For example, Chinook salmon stock proportions were similar among floodplain wetland and main-stem sampling sites in the lower Willamette River (Teel et al. 2009) and between interior wetland channel and main-stem habitats in adjacent areas of the lower estuary (Bottom et al. 2011). Substantial differences in stock composition become apparent, however, when deep mid-channel habitats are also surveyed. For example, purse seine collections from mid-channel habitats in the lower estuary include higher proportions of interior spring and fall run stocks and lower proportions of lower Columbia River fall Chinook stocks compared to near-shore sites sampled with the beach seine (Roegner et al. 2008, Weitkamp et al. 2012). The higher prevalence of interior spring stocks (which migrate primarily as yearlings) in purse seine collections is consistent with observed size-dependent patterns of estuary habitat use and migration: yearlings and large subyearlings are generally more abundant in deep habitats, whereas fry and fingerlings dominate in wetland channel and near-shore areas.

In March 2010, NOAA Fisheries initiated a 2-year series of bimonthly genetic surveys to compare shallow-habitat stock distributions at an estuary-reach scale and to ensure that the patterns previously observed are not merely an artifact of different study periods, locations, and methods. The surveys targeted three shallow habitat types in the tidal-fluvial hydrogeomorphic reaches (C–E) between ~rkm 100 and Bonneville Dam. The preliminary results support the general patterns described above (D. Teel, Northwest Fisheries Science Center, unpublished data) and reinforce the conclusion that variations in stock composition are consistent between years and at the estuary-reach scale. Final results of the 2-year genetics surveys will be reported by the fall of 2012.

4.1.2 Stock-Specific Life Histories and Performance

Variations in migration timing and sizes reflect a diversity of juvenile life histories within and among the genetic stock groups surveyed in shallow estuarine habitats (Roegner et al. 2012). Whereas West Cascade Tributary fall and Spring Creek group fall stocks in the lower estuary are represented primarily by fry and fingerling migrants, less abundant stock groups, including West Cascade tributary spring and Willamette River spring stocks, encompass a wider range of sizes and ages, including subyearling and yearling migrants. Recent lower Willamette River surveys (Teel at al 2009) similarly noted that spring Chinook salmon stocks produce not only yearlings, but also fry and fingerling migrants that use shallow tidal habitats before entering the ocean.

Otolith studies indicate that juvenile Chinook salmon in shallow-water habitats near the estuary mouth include a high proportion of individuals that had reared in the estuary prior to capture (Bottom et al. 2011). Approximately 80% of 192 juvenile Chinook salmon otolith samples collected from Point Adams Beach and analyzed for life history in 2004 and 2005 were classified as Spring Creek group and West Cascade Tributary fall stocks. However, a diversity of other stocks were represented, including Upper Columbia River summer/fall (n = 9 in 2004) and introduced Rogue River stocks (n = 11 in 2005). Estimated residence times in the brackish portion of the lower estuary ranged from 1 month to several months prior to capture. Back-calculated size at saltwater entry averaged ~60 mm for the Spring Creek group and West Cascade Tributary fall stocks, and their annual mean residency times ranged between 24 and 76 d for the 2 years. Despite a higher estimated mean size at estuary entry (~88 mm FL), residency estimates for upper Columbia River fall summer/fall stocks averaged 82 d. These residence time results represent minimum values because the otolith technique does not account for days of residence in the large tidal-fresh portion of the estuary or for any additional days that captured individuals might have remained in the estuary before entering the ocean.

Although salmon residence time in the estuary tends to vary inversely with fish size (Campbell 2010), monitoring of PIT-tagged fish in shallow habitats of the lower estuary also suggest that even some hatchery-reared salmon may linger in the estuary for weeks before migrating seaward. In 2009, PIT antennas deployed in a shallow wetland channel at Russian Island detected 17 Chinook salmon released at Bonneville Dam or at Spring Creek National Fish Hatchery. Travel times to the lower-estuary wetland averaged ~41 d (21-61 days), indicating that at least some hatchery fish migrated slowly and entered shallow wetland channels in the lower estuary (Bottom et al. 2011). This finding differs from the results of acoustic tagging studies that reported Chinook salmon migrating through the entire estuary (Bonneville Dam to near the estuary mouth) within 3 or 4 d (McComas et al. 2008; McMichael et al. 2010). Estimated estuary residence times for Chinook salmon thus vary widely depending on the experimental methods chosen and the particular habitats, stocks, and life histories targeted by each study.

Recent tagging studies in the Sandy River delta provide evidence of unexpected diversity in the estuary rearing behaviors of multiple stocks of Chinook salmon. From January through April 2010, G. Johnson et al. (2011a) captured and tagged fish in the Sandy River delta and monitored their local residency based on tag detections at acoustic receivers. Nearly 60% of the 51 tagged fish were classified as Willamette River spring Chinook salmon and most likely originated from stock transfers into the Sandy River basin. Most of the other tagged fish were West Cascade Tributary (spring and fall) stocks, but several upriver stocks—Snake River spring and fall and upper Columbia River summer/fall—also were represented. For 48 tagged individuals (mean FL = 111 mm) with at least one valid detection, residence time in the area averaged 34 d. One-quarter of the locally tagged fish were fall Chinook stocks that may have over-wintered in shallow areas of the tidal-fluvial estuary rather than enter the ocean during their first year of life (G. Johnson et al. 2011). The extended residency of locally tagged Chinook salmon during the winter/spring period contrasts sharply with monitoring results for other groups tagged above Bonneville Dam that migrated rapidly through the Sandy River delta and vicinity during spring and summer.

Burke (2005) hypothesized that life history diversity of juvenile Chinook salmon within the estuary has declined relative to the complex patterns described from a series of surveys and scale-pattern analyses in 1914-16 (Rich 1920). Results of lower-estuary surveys in 2002-08 further support this hypothesis: assuming Rich's (1920) results are representative of historical life histories, far fewer subyearling migrants now enter the estuary during summer and fall than did a century ago. Multiple factors could

explain the apparent shift, including reduced rearing opportunities within the estuary, population losses from upriver stocks that formerly contributed late migrants to the estuary, and hatchery programs that have concentrated production of relatively few salmon phenotypes (Bottom et al. 2005, 2011).

Although a few RME studies have documented variations in estuary residency and life history diversity within stocks, most salmon performance measures—e.g., foraging success, growth, condition—depict the average result for all stocks occupying a habitat or reach. Sample sizes are often too small to quantify performance by stock group, particularly for those at-risk stocks that are poorly represented in field collections. This is further complicated by analytical methods that require subsampling. For example, otolith techniques for life history and growth determinations are costly and require lethal sampling, limiting the number of individual fish that can be collected and processed from any one stock group. Through long-term monitoring, sample sizes for poorly represented stocks might be increased sufficiently to more adequately characterize their estuary habitat use and performance.

4.1.3 Estuary Contributions to Population Recovery and Viability

RME studies to support salmon recovery have primarily explored salmon ecology and performance within the estuary, particularly in shallow-water habitats. The results indicate that most if not all Chinook stocks are able to express a diversity of life histories, including extended periods of estuarine rearing (i.e., weeks or months) before migrating to sea. Empirical studies have shown relationships between habitat use in the estuary and juvenile performance, however, they have not quantified the importance of estuary performance to population viability or the benefits of restoring estuarine habitat opportunities for salmon recovery. This implies a fundamentally different research approach that places the estuarine life histories of salmon in a life-cycle context. Many studies in the basin have adopted elements of a life-cycle approach, particularly in freshwater areas, near the source of known spawning populations. However, downstream in the estuary populations throughout the Columbia basin intermix, and the natal population sources for most individuals may be impossible to identify. Recent studies have begun to use otolith micro-chemical techniques to identify the juvenile life histories represented among Chinook spawners returning to selected ESUs. Life-cycle models are also being used to explore the sensitivities of salmon populations to improvements in estuary performance and subsequent ocean survival. Both otolith and modeling studies are progressing but published results were not available at the time of this review.

Tagging experiments have been widely used in the Columbia River as a method for estimating the adult population response to various management strategies for improving salmon survival at particular life stages. For example, a large number of studies have estimated the survival of selected stocks with different migration pathways (i.e., barge transport vs. in-river migration) through the hydrosystem to assess “delayed mortality” effects at subsequent life stages (e.g., Muir et al. 2006; Schreck et al. 2006). The presence of PIT detectors at each main-stem dam allow survival to be estimated for tagged out-migrants and returning adults throughout the riverine migrations of various stocks. In many of these studies estuarine and marine mortality are simply lumped as a single ocean-to-adult survival value because no population counts are made between the time a smolt passes Bonneville Dam and returns as an adult (Haeseker et al. 2012). To quantify mortalities within the estuary, some studies have used acoustic receivers or radiotelemetry to monitor smolt movements and estimate mortalities at various locations below Bonneville Dam (Schreck et al. 2006; Welch, et al. 2008; Clemens et al. 2009). Experimental hatchery groups released into the estuary also have been used to examine the effects of migration timing (e.g., Muir and Emmett 2008) and size (e.g., Claiborne et al. 2011) on ocean survival and adult returns.

Beyond estimating survival, an important advantage of PIT- or acoustic-tagging methods is the ability to monitor variations in migratory and rearing behaviors that may contribute to adult returns and benefit population resilience (Bottom et al. 2005; Healey 2009). For example, using PIT-tagging methods and scale analysis, Connor et al. (2005) concluded that Snake River fall Chinook salmon express at least two alternative juvenile life histories that contribute to total adult returns: ocean-type (subyearling migrants) and reservoir-type (yearling migrants). This finding contradicts historical data indicating that the population consisted almost entirely of subyearling migrants, and implies that a novel life history adaptation has developed in response to the storage reservoir system (Williams et al. 2008). However, scale pattern analysis could not discriminate between adults that had spent their first winter in a reservoir and those that may have resided somewhere below Bonneville Dam (Connor et al. 2005). Subsequent PIT analyses have confirmed that a significant number of transported Snake River subyearlings indeed overwinter below the dam (Marsh et al. 2010). Experimental tagging recently identified a variety of lower and upper Columbia River fall Chinook stocks—including Snake River, Spring Creek, Upper Columbia, and West Cascade Tributary groups—that overwinter in the upper estuary near the Sandy River delta (G. Johnson et al. 2011).

Unfortunately most tagging methods are not suitable for tracking small subyearling migrants in the Columbia River estuary or monitoring their relative contributions to returning adults. Acoustic tags are too large to tag salmon <~95mm, and although PIT tags can be applied to fish as small as 60 mm, a large proportion of the juveniles in tidal wetland channels consists of smaller fry migrants (Bottom et al. 2011). The narrow detection range of PIT antennas further limits the habitat types and spatial scales that can be studied using remote PIT-detection systems. Other research approaches are needed to account for the full diversity of size classes and migratory patterns expressed by many populations.

Small estuary tributaries may provide one useful alternative for investigating the importance of estuarine habitats to the viability of individual populations. Such tributaries may provide a useful small-scale analog for understanding salmon habitat use in the main-stem estuary, including (1) a full continuum of freshwater-tidal habitats, where the juvenile life histories of a known population can be quantified; and (2) an associated network of freshwater rearing and spawning habitats, where river flows and the migrations of salmon are unimpeded by main-stem dams. For example, Craig (2010) used salmon scales to classify the juvenile life histories of a coho salmon population in Grays River, a tributary of the lower Columbia River estuary. Through smolt trapping upriver and frequent field collections in the estuary, she documented juvenile coho migration times, sizes, and distributions and identified consistent scale patterns corresponding to particular migrant and rearing behaviors. She identified at least five prominent juvenile life histories in the Grays River population. The results provide a catalogue of scale patterns for at least one coho population that could be analyzed in adult scales to quantify the relative contribution of each juvenile life history to returning adults.

Among interior salmon populations that enter the main-stem Columbia River far upstream of the estuary, alternative methods may be required to interpret the estuary's contributions to adult returns and population viability. Campbell (2010) compared the effectiveness of scale morphometrics, scale chemistry, and otolith chemistry for reconstructing the juvenile life histories of various stocks of Chinook salmon sampled in the LCRE. He concluded that scale morphometrics does not provide a consistent indicator of estuary entry for classifying the juvenile life histories of Chinook salmon. Otolith strontium offered the most reliable and sensitive indicator of Chinook entry into the saline portion of the estuary. However, because no suitable chemical indicator has been established for the tidal fresh environment, the otolith technique can only provide a minimum estimate of estuary residency. Thus, otolith chemistry

alone may not be a satisfactory tool for life history studies involving stocks that reside primarily in the upper estuary.

The results to date indicate that no single research tool or design will be adequate to interpret the estuarine life histories or quantify the estuary's contributions to all stocks. A combination of approaches specific to the sampling challenges and life histories of each ESU may be required.

4.2 Conclusions

Until recently, fish surveys in the LCRE provided general descriptions of the distribution and abundance of juvenile salmon. The upriver sources of estuary-rearing salmon could only be determined for individuals that had been tagged in their natal basins or in hatcheries and later recaptured. Not surprisingly, information about stock-specific rearing and migration behaviors was based primarily on results from relatively large, tagged hatchery smolts. In the last decade, new tagging techniques, otolith chemical analyses, and an improved genetic baseline for Chinook salmon have greatly expanded our capabilities for interpreting stock-specific patterns of estuary rearing and migration. Genetic results have documented variations in the stock composition of Chinook salmon in various estuary reaches and habitats. Tagging studies and otolith chemical methods have described life history variations for a few genetic stock groups. Overall, limited results to date suggest that estuary residency and habitat use vary among stocks and their associated entry locations, times, and sizes. These findings have important implications for selecting estuary restoration projects more strategically to satisfy the diverse estuary migration pathways and habitat requirements of salmon from different ESUs.

Despite a wealth of new data about stock-specific habitat use, life histories, and performance of juvenile salmon in the estuary, much remains to be learned about the importance of estuary rearing to population viability and salmon recovery. Continued estuary monitoring is needed to more fully characterize juvenile life history variations within and among genetic stock groups, including at-risk stocks that are in low abundance and often poorly represented in estuary sample collections. Mid- and upper reaches (D – H) of the estuary have been surveyed less intensively than those in the lower estuary. Additional surveys will be required in this region to encompass the full range of habitat types or time periods for different genetic stock groups. Most RME studies have targeted shallow-water and near-shore areas, including habitat types that have been most intensively modified by historical development and that are the primary focus of estuary restoration. Methods for sampling deeper channels further from shore (e.g., purse seine, pair trawl, acoustic–tag monitoring, etc.) often select for high proportions of yearlings and hatchery fish that tend to move most rapidly through the estuary during punctuated migration periods. Additional surveys in deep channel habitats may be useful if the objective is to estimate survivals or migration rates for rapidly migrating stocks (e.g., chum, steelhead, sockeye) or to compare stock-specific life histories (i.e., subyearling and yearling migrants) across a wider range of estuarine habitat types.

Most RME studies have evaluated salmon habitat use or performance within the estuary and have not determined whether estuary rearing conditions influence adult survival. New life-cycle approaches to research and monitoring are needed to quantify the estuary's linkages to salmon populations and to evaluate the importance of estuarine habitat opportunities for salmon recovery. A series of indicator populations and experimental methods should be employed to directly measure the contribution of estuarine habitats to adult returns and population viability.

5.0 Estuary Restoration Actions and Salmon Performance Within the Estuary

Tracking restoration projects and associated action effectiveness (AE) research within the LCRE proved to be a formidable task because numerous federal, state, and local entities participate in various phases of ecosystem restoration and research. According to the Lower Columbia River Estuary Partnership (LCREP) database, 42 confirmed aquatic restoration projects (e.g., hydraulic reconnections, channel creation, large woody debris [LWD] placement) have restored a total of 3152 acres since 2001. If land acquisition and non-aquatic based restoration (e.g., re-vegetation, invasive control) are considered, the number of projects in the LCRE and surrounding tributaries equals 93 and totals 6294 (K Marcoe, personal communication, 3 July 2012). Restoration activities in the LCRE have included a variety of actions ranging from riparian and vegetation planting, tide-gate replacement and/or removal, dike removal and/or breaching, as well as excavation and creation of shallow-water habitats.

Much of the written material relevant to AE research is typically conveyed in annual report format as part of contractual requirements to funding entities. Because funding sources for restoration in the LCRE vary, information relevant to AE research and evaluation was not readily accessible, which proved to be problematic when attempting to establish a systematic approach for obtaining material to review. In lieu of traditional literature review methods (e.g., databases), AE reports were obtained by coordinating directly with the funding entities. During this process, it became clear that while there have been numerous restoration activities in the LCRE over the past decade, very few have included AE monitoring. Of the projects that included before and/or after restoration monitoring, some monitored structural features and/or physical conditions, but few examined metrics that were directly linked to juvenile salmon performance (e.g., realized function; see section 3.4). For the purpose of evaluating the link between restoration activities and salmon performance, projects that included fish sampling as part of their respective investigations were reviewed. Nine projects met this criterion (Table 5.1); their locations are pinpointed on the LCRE images included in the following sections. All of the reviewed AE projects involved restoration activities that dealt with hydraulic reconnections and/or improvements, and all of these evaluations examined attributes associated with capacity, opportunity, and/or realized function (i.e., salmon performance). The review that follows emphasizes juvenile salmon response to restoration actions, as opposed to ecosystem controlling factors, structures, or processes.

Table 5.1. Action Effectiveness Research in the Lower Columbia River and Estuary

Site Name	River km	Restoration Action
Fort Columbia	14	Culvert replacement, channel excavation, large woody debris placement
South Slough	19[(a)]	Tide gate replaced with bridge
Vera Slough	19[(a)]	Tide-gate replacement
Kandoll Farm	37[(a)]	Tide gate replaced with culvert, dike breach
Tenasillahe	56	Tide-gate replacement
Julia Butler Hansen	58	Tide-gate installation and replacement, culvert repair
Crims Island	90	Excavation of marsh elevation, channel creation
Hogan Ranch	140[(a)]	Water-control structures, cattle exclusion fencing, invasive plant removal, native replanting.
Mirror Lake	208	Culvert replaced with a bridge, riparian restoration, LWD enhancement, culvert improvements

(a) River kilometer for sites not directly adjacent to the main-stem of the Columbia River were approximated based on the general vicinity of the site relative to the main-stem.

5.1 Fort Columbia (rkm 14)

The Fort Columbia site includes 96 acres of wetlands and is located near the town of Chinook, Washington. Historically, the wetland drained into the Columbia estuary, but road construction during the 1950s diminished hydraulic connectivity at this site by installing a 24-in. perched culvert. Restoration occurred during 2011 by means of replacing the 24–in. culvert with a 12-ft by 12-ft box culvert and excavating a tidal channel to reconnect the wetland. Habitat complexity was established by adding LWD to the excavated channel. Pre-restoration data were not collected in the wetland channel because site conditions were not suitable for sampling fish (CREST 2011c). It does not appear that a reference site was established to accompany AE monitoring.

Post-restoration data collected within the wetland channel at Fort Columbia indicate juvenile salmon are accessing the site. Construction was completed in February 2011 and, at the commencement of the first post-restoration sampling event the following month, Chinook and coho salmon were found at the site. Aside from counts of species found during monitoring efforts in 2011, CREST (2011c) reported no other data; therefore, inferences about the effectiveness of restoration are limited at this time. The inclusion of other metrics (e.g., sizes, condition factors, water quality) and reference sites would strengthen the ability to infer the effects of restoration actions on the performance of juvenile salmon.

5.2 South Slough (rkm 19[1])

The South Slough site includes 45 acres of diked pasture and is located in the Lewis and Clark National Historic Park, near the town of Warrenton, Oregon. The South Slough drains into the Lewis and Clark River, which ultimately discharges into Young's Bay before reaching the main-stem of the Columbia River. Restoration actions involved replacement of a failing culvert with a bridge to increase hydraulic connectivity. The bridge was installed during 2007. A before-after-control-impact (BACI) design was implemented to examine the ecosystem response to restoration. Pre-restoration data were collected during 2007 and post-restoration data generally were collected from January through August 2008 to 2011, although sampling effort varied across months and years (CREST 2012a).

[1] River kilometer for sites not directly adjacent to the main stem of the Columbia River were approximated based on the general vicinity of the site relative to the main stem.

Sampling gear and effort varied across years and between the reference and restored site, which makes a BACI analysis for fish abundance improbable. Across sampling years, five species of juvenile salmon and trout were captured at the restoration and reference sites; coho and Chinook salmon were the most abundant. The total number of juvenile Chinook salmon captured during the 5-year survey period ranged from 0 to 19 individuals per year. The range of Chinook salmon sizes was indicative of multiple life history strategies using the marshes at Fort Clatsop in both the restored and reference sites. Condition factor indices indicated slightly higher values for salmon in the restored site than in the reference site. Species diversity as measured by the Shannon-Weiner index was higher in the reference site than in the restoration site across all years. Non-native taxa were captured at both the restored and reference sites and accounted for approximately 20% of the catch at both locations (CREST 2012a).

Results from measurements of water properties indicate a response in some metrics that corresponds to restoration actions. Measurements of water depth indicate the South Slough site shifted toward an increase in tidal amplitude after restoration actions. The 7-day moving average temperature at both the reference and the restored sites either approached or exceeded 16°C during late summer months. After restoration, maximum water temperatures at South Slough were lower compared to pre-restoration data. Maximum water temperatures at the restored site were also lower than the reference site; this trend was attributed to a combination of habitat conditions (e.g., greater water depth and freshwater input) (CREST 2012a).

The disparity in consistent sampling across years and between the reference and restored sites hampers opportunities to conduct analyses aimed at articulating salmon response to restoration actions. The low sample sizes of juvenile Chinook salmon captured during this study may further limit the potential to make inferences about salmon performance in the restored channel. CREST (2012a) provided some notable observations related to reference site selection. The reference site was found to be quite different from the restoration site in terms of habitat complexity. Comparatively, the reference site had a smaller channel that retained less water and had no upland freshwater input. These conditions were speculated to have yielded different results in water properties (CREST 2012a), but these conditions may also lead to differences in habitat opportunity and capacity between restored and reference conditions. These findings demonstrate the need for careful consideration of habitat attributes when selecting reference sites that are to be used for AE evaluations.

5.3 Vera Slough (rkm 19[1])

Discharging into Youngs Bay, Vera Slough is adjacent to the Astoria Airport and near the town of Warrenton, Oregon. The wetland- and shrub-dominated characteristics that existed historically were converted into farmland and developed for the airport. The site is presently characterized as a brackish marsh. Restoration at this site occurred between 2005 and 2006 and involved a tide-gate retrofit. Pre-restoration data were collected during

[1] River kilometer for sites not directly adjacent to the main stem of the Columbia River were approximated based on the general vicinity of the site relative to the main stem.

2005 and post-restoration data were collected during 2006. Data collection efforts occurred inside of the restoration site, outside of the tide-gated site, and at a reference site (G. Johnson et al. 2011).

In contrast to the immediate hydrologic effect at the restored Kandoll Farm site, the tide-gate replacement at Vera Slough did not result in water-surface elevations that mimicked reference site conditions, although the restoration action did increase the tidal amplitude into the site (G. Johnson et al. 2011). During spring and summer months, water temperature inside the tide gate at Vera Slough was warmer compared to the reference site. Furthermore, the 7-DAM water temperature frequently exceeded the 16°C criterion for salmon.

There were few differences in fish community characteristics during the pre- and post-restoration periods inside Vera Slough, outside Vera Slough, and at the reference site. Prior to restoration, species richness and diversity was lower inside Vera Slough than in areas outside of the tide-gated and reference sites. Post-restoration, species richness and diversity increased compared to pre-restoration conditions. However, the fish community composition did not approach diversity and richness levels comparable to nearby areas on the main-stem Columbia River. Catches of salmon at Vera Slough were notably low both pre- and post-restoration.

Based on the initial AE data collection efforts, the Vera Slough retrofit tide-gate project appeared to have little benefit to juvenile salmon the year after restoration. Without multiple years of post-restoration data and multiple monitored metrics by which to infer benefits to salmon performance it is difficult to conclude the response trajectory of this particular project.

5.4 Kandoll Farm (rkm 37[1])

Located along the floodplain of Grays River (rkm 2.5), Kandoll Farm was historically a Sitka spruce swamp, but was converted to wet pasture. Restoration at this site occurred between 2005 and 2006 and involved tide-gate removal, culvert installation, and dike breaching. The AE research at Kandoll Farm included pre-restoration monitoring as well as reference sites (Roegner et al. 2010). Restoration actions had an immediate effect on the water inundation at Kandoll Farm. Water-surface elevations emulated tidal ranges similar to reference sites and water temperatures were more aligned with those in the reference sites (G. Johnson et al. 2011)

During the season preceding tide-gate removal at Kandoll Farm, the only taxon captured inside the site was threespine stickleback. In contrast, the reference site yielded seven species of fish. After restoration actions at Kandoll Farm, nine species of fish, including three species of salmon, were sampled from the restoration site. Chinook salmon were infrequently captured compared to chum and coho

[1] River kilometer for sites not directly adjacent to the main stem of the Columbia River were approximated based on the general vicinity of the site relative to the main stem.

salmon. In terms of life stage, the Chinook salmon captured at the site were predominantly fry and fingerlings, and few yearlings. The chum salmon were nearly all fry-sized, and coho salmon were represented by fry, fingerlings, and yearlings (Roegner et al. 2010; Johnson et al. 2011). Juvenile salmon diets were more diverse in the restored wetland than in the Grays River. Overall, prey items accounted for in the diets of juvenile salmon were dominated by insects that were presumably derived from wetland habitats.

Restoring hydraulic connectivity at Kandoll Farm increased the opportunity for fish to access the site. This was noted by an increase in species richness and diversity (Roegner et al. 2010, G. Johnson et al. 2011). The higher abundance of chum salmon and coho salmon at Kandoll Farm compared to that of Chinook salmon is notable for several reasons. First, this trend differs from the findings of several AE research projects in the LCRE that report Chinook salmon to be the most abundant juvenile salmon species captured in shallow-water sites (Johnson et al. 2008, Haskell and Tiffan 2011, Sagar et al. 2011). Second, this pattern indicates local conditions as well as the status of nearby populations of salmon that may access and benefit from restoration actions are worthy of considerations for prioritizing site-scale restoration within the LCRE.

5.5 Tenasillahe Island (rkm 56)

Tenasillahe Island is located at rkm 56 and is downstream of Puget Island and the town of Cathlamet, Washington. During the summer of 2007, the USACE replaced three top-hinge steel tide gates on Tenasillahe Island with side-hinged aluminum tide gates. AE research was conducted to evaluate the effects of the restoration activities on juvenile salmon. The AE research was designed to perform before and after restoration comparisons with treatment (Tenasillahe Island) and reference (Welch Island; rkm 55) sites. Pre-restoration data were collected from March through June 2006 and March through May 2007 (J. Johnson et al. 2008).

From March through July 2008, the new tide gates were estimated to have opened for approximately 4.4 hours per d. J. Johnson et al. (2008) did not capture any salmon entering the Tenasillahe Island slough. A total of 27 Chinook salmon were captured emigrating from the site and at least 12 of these Chinook salmon were part of tag-and-release experiments conducted during the 2008 research on the island. Hatchery reared Chinook salmon were used to investigate residence time in Tenasillahe Island by using a combination of marking techniques: fin clips and PIT tags. The residence time of marked fish within the channels at Tenasillahe Island ranged from 1 to 119 d with a median that ranged from 41 to 45 d for the various tag groups. Growth rates of recaptured PIT-tagged Chinook salmon at Large Tenasillahe Slough ranged from 1.29 to 1.62 mm/d (J. Johnson et al. 2012).

Community composition between restoration and reference sites was investigated using a combination of beach seines and hoop nets. J. Johnson et al. (2008) found a higher proportion of native taxa in reference sloughs on Welch Island than at the treatment sloughs on Tenasillahe Island. Chinook

salmon were the most abundant of the salmon species captured. However, very few Chinook salmon were captured at the restoration sites on Tenasillahe Island (n=2), compared to the reference sites on Welch Island (n=229) (J. Johnson et al. 2008).

In addition to fish community composition, differences between restoration and reference sloughs were noted in water temperature and DO concentrations. From March through June 2008, the 7-DADM exceeded 16°C for 47 d at the Tenasillahe Island sloughs and for approximately 25 d at the Welch Island sloughs. In terms of temperature range, the gated sloughs on Tenasillahe had a lower range in daily temperature compared to the reference sloughs on Welch Island. While there was a broad range in DO concentrations between reference and restored sites, the lowest DO concentrations were noted in the gated sloughs on Tenasillahe Island (J. Johnson et al. 2008).

The AE study design of the Tenasillahe Island tide-gate replacement project included before monitoring and use of a reference site. A BACI analysis was not included in the 2008 progress report. Based on post-restoration results, it seems that few salmon enter the Tenasillahe Island tide gates. The opportunity for access by juvenile salmon and other fish is limited to discrete time periods when water elevation in the slough exceeds water elevation in the river, which occurred less than 20% of the time on a particular day (J. Johnson et al. 2008). In addition, the capacity of the tide-gated sloughs to support salmon appears to be limited based on water-quality conditions at the restoration site compared to the reference site; warmer temperatures and lower DO concentrations were noted at the restoration site in comparison to the reference site.

5.6 Julia Butler Hansen National Wildlife Refuge (rkm 58)

Encompassing over 5600 acres and located near the town of Cathlamet, Washington, the Julia Butler Hansen National Wildlife Refuge (JBH) is primarily managed for endangered Columbian white-tailed deer. Habitats within the refuge are varied and include pastures, wetlands, and forested tidal swamps. During the summer of 2009, the USACE repaired and replaced tide gates at two sloughs. In addition, two sloughs that had been previously disconnected via dikes were hydraulically reconnected with newly installed culverts and tide gates. AE research was conducted under a BACI framework, which included pre-restoration monitoring and the use of reference sites. Pre-restoration AE research occurred during spring of 2007 and 2008 and post-restoration research occurred during spring 2010 (J. Johnson et al. 2009, 2011).

Prior to restoration activities, juvenile salmon were captured entering reference sloughs as well as tide-gated sloughs, although the proportion of juvenile salmon captured at the reference sloughs was higher compared to the tide-gated sloughs. A higher proportion of larger (>110 mm) juvenile Chinook salmon were captured in the tide-gated sloughs compared to the proportion of Chinook salmon captured within the reference sloughs. The data also suggest the tide-gated sloughs were not as accessible to

smaller sized (>65 mm) Chinook salmon. Tide-gated sloughs and blocked channels also had a higher proportion of non-native taxa compared to reference sites (J. Johnson et al. 2009).

After restoration, more juvenile salmon (species as well as abundance) were captured entering the newly connected tide-gated slough than entering the reference slough. Juvenile Chinook salmon were the most abundant salmon species captured in both restored and reference sites. Coho salmon were captured in both restored and reference sites, but chum salmon and steelhead occurred only in the restored, tide-gated slough. Similar to the trends observed prior to restoration, the size distribution of juvenile Chinook salmon captured in the tide-gated sloughs was larger, with a higher proportion of fish >65 mm, compared to sizes of fish in reference sites. After restoration, and similar to pre-restoration findings, tide-gated sloughs had a higher proportion of non-native taxa than reference sloughs (J. Johnson et al. 2011).

There was no significant difference in DO concentrations between the reference and tide-gated sloughs after restoration actions. Water temperature varied through space and time. After restoration, from April through June, the 7-day average daily maximum (7-DADM) exceeded 16°C in the two reference sloughs for 44 and 16 d, respectively. In the three tide-gated sloughs, the 7-DADM exceeded 16°C for 32, 24, and 6 d, respectively. Depending on the particular site, the 16°C threshold was exceeded during various times during the 3 months monitored—April, May, and June. The reference sites yielded the highest median daily temperature ranges (J. Johnson et al. 2011).

Juvenile salmon were accessing tide-gated sloughs at JBH before and after restoration. Furthermore, restoring connectivity to previously blocked channels by installing tide gates appears to have created opportunity for fish access to the sites, although in some cases access already existed via drainage ditches. Regardless, densities of salmon were higher in unrestricted reference sloughs than behind tide-gated channels. The magnitude of these changes within the context of response to restoration actions has yet to be determined. Johnson J. et al. (2011) note an analysis of pre- and post-restoration with regard to fish passage will be forthcoming.

5.7 Crims Island (rkm 90)

Crims Island is located along the Columbia River main-stem at rkm 90. The goal of the Crims Island restoration project was to improve habitat for juvenile salmon as well as for Columbian white-tailed deer. Restoration occurred between 2004 and 2005 and involved excavation of the marsh surface elevation and digging channels within the island. Pre-restoration data were collected in 2004 and post-restoration data were collected during spring and summer months from 2006 through 2009. A reference site was established at a nearby location, Gull Island (rkm 89). Statistical analyses were applied under the before-after comparison framework to examine the size of juvenile Chinook salmon, condition factor, corrected prey weight, invertebrate density, and diversity. The abundance of fish could not be evaluated in the before-after comparison, because physical

changes at the site resulting from restoration activities resulted in the need to adapt to the new conditions by changing sampling locations as well as sampling techniques (Haskell and Tiffan 2011).

After restoration activities, the mean size of subyearling Chinook salmon was 59 mm, which was approximately 10 mm larger than the pre-restoration conditions and represented a significant increase (P=0.007). In addition, restoration appeared to be correlated with an increase in the condition factor for juvenile Chinook salmon. Overall, the abundance of Chinook salmon was noted to decrease around June, which also corresponded to a time during which water temperatures exceeded 20°C. The densities of Chinook salmon were highest in subtidal channels, followed by intertidal, and finally marsh plains; although the differences were not statistically significant. Chinook salmon in subtidal channels were significantly ($P<0.05$) larger than fish in the intertidal channels (Haskell and Tiffan 2011). In addition to the abundance, size, and distribution of juvenile Chinook salmon at the Crims Island site, Haskell and Tiffan (2011) describe patterns in feeding behavior, residence time, and invertebrate characteristics associated with pre- and post-restoration conditions. Feeding behavior was variable in space and time for juvenile Chinook salmon at Crims Island. Dipterans were common prey items at both the restoration and the reference sites. The diets of Chinook salmon at main channel sites yielded different feeding behaviors because diets here were dominated by *Corophium* spp. and *Daphnia* spp. While the median residence time of Chinook salmon was noted to increase after restoration, overall residence times were abrupt (e.g., hours) (Haskell and Tiffan 2011). There were no significant differences in the densities of benthic and drift invertebrates between pre- and post-restoration, although the diversity of these prey pools decreased after restoration, and in some cases this decrease was found to be statistically significant (Haskell and Tiffan 2011).

The effectiveness of the Crims Island restoration project was thoroughly evaluated by making statistical comparisons of selected metrics before and after restoration and within the context of a nearby reference site. The inclusion of appropriately selected attributes related to juvenile salmon ecology (e.g., habitat selection, feeding behavior, prey densities, residence time) within the study design facilitated a holistic understanding of juvenile salmon response to restoration actions. Mean size of Chinook salmon was significantly larger after restoration. It is unknown if this change was related to an increase in the capacity of the system which positively affected foraging success, or was a result of increasing opportunity for larger size classes of fish. Findings related to different densities and sizes of Chinook salmon across different habitat types (e.g., subtidal and intertidal channels) may offer insight to the design criteria for restoration sites in the LCRE. The short residence times reported by Haskell and Tiffan (2011) were much shorter than those reported by others in the LCRE (see Bottom et al. 2011, Johnson et al. 2011). However, given that residence time was investigated during a single event in May at Crims Island, it is plausible that an expanded effort aimed at targeting different life history types and/or additional time periods may yield different results. The finding of no differences in the densities of invertebrate prey pools between pre- and post-restoration conditions was notable, and is likely worthy of further consideration for future AE research at other locations in the LCRE.

5.8 Hogan Ranch (rkm 140[1])

Situated between Scappoose Bay and the Multnomah Channel, Hogan Ranch is part of the Scappoose bottomlands and is characterized by a complex network of wetland channels and ponds that ultimately connect to Scappoose Bay. Initial phases of restoration at the Hogan Ranch site began 2005 when fencing was installed to exclude livestock. During 2007, the exclusion fencing was completed and in-water work commenced to replace failed water-control structures. Additional activities included dike removal, excavation to create wetland habitat, and vegetation planting (Sagar et al. 2011). The water-control structures act as barriers for fish passage during low-water periods and may also prevent escapement if fish gain access to the site during high-flow conditions (CREST 2011a). Monitoring for this project occurred after restoration actions, from 2008 through 2010, and was conducted to understand whether juvenile salmon were accessing the site and to determine if stranding was an issue. Juvenile salmon diet and prey were also evaluated. The sampling design did not appear to include a BACI design or a reference site.

During the 3 years of spring-summer sampling at Hogan Ranch, a total of eight juvenile salmon were captured: three Chinook salmon and five coho salmon; all were unmarked. No salmon were captured in the ponds via seining. The salmon encountered at Hogan Ranch were captured using a fyke trap at Teel Slough, a tidally connected channel to Scappoose Creek. CREST (2011a) speculated the lack of salmon in the ponds may have been attributed to site conditions (e.g., high density of aquatic vegetation), which may have rendered beach seining ineffective. Furthermore, seining in the ponds was limited to June and July when water temperatures are typically their warmest in shallow-water habitats. The abundance of invasive fish species at Hogan Ranch was attributed to a shallow-water with little riparian cover as well as limited flow velocities. Teal Slough was also noted to have high water temperatures; however, increased connectivity with nearby water bodies may have resulted in water-quality conditions that were more favorable to supporting a higher diversity of species, compared to the pond habitats (CREST 2011a).

From 2007 through 2010, a suite of water properties was examined at Hogan Ranch during spring and summer months to evaluate the effects of cattle exclusion and riparian re-vegetation on wetland functions. Results indicate few consistent trends between sites and years. *E. coli* was the single parameter that yielded the most response through time; levels decreased in all ponds. Water temperature was generally greater than 18°C from spring through early fall (Sagar et al. 2011). Based on the multiyear results of water properties at Hogan Ranch, Sagar et al. (2011) concluded that aquatic habitats at this site were not amenable to supporting juvenile salmon.

The effectiveness of the restoration actions specific to juvenile salmon cannot be determined due to extremely low capture rates of these taxa. As noted by Sagar et al. (2011) the elevated temperatures

[1] River kilometer for sites not directly adjacent to the main stem of the Columbia River were approximated based on the general vicinity of the site relative to the main stem.

appear to diminish opportunity for juvenile salmon to access the site during summer months. It is unknown how or if juvenile salmon access and gain benefit from this site during other time periods. Because of reported gear inefficiencies in the pond habitats it is unknown whether salmon stranding occurs at Hogan Ranch. The lack of pre-restoration data and a reference site hinders the opportunity to attain a holistic understanding of salmon performance as a result of the restoration actions at Hogan Ranch.

5.9 Mirror Lake (rkm 208)

Within the bounds of Rooster Rock State Park, Mirror Lake is segregated from the main-stem of the Columbia River by Interstate 84 near rkm 208. Restoration activities at Mirror Lake began in 2005 when a failing culvert was replaced with a bridge. Restoration activities occurred over a number of years and included such actions as, riparian planting, placement of LWD, and culvert improvements aimed at facilitating fish passage (Sager et al. 2011). A BACI design was not applied, nor were there clearly defined reference sites. Therefore, the ability to make inferences about the response of salmon to the restoration activities at Mirror Lake is limited. However, metrics associated with habitat opportunity and capacity for juvenile salmon were examined in the context of other monitored locations and may serve, for comparative purposes, as reference sites.

Jones et al. (2009) describe thermal patterns linked to differing habitat types. The lower portions of the site were characterized as shallow wetland habitats experiencing little flow, which resulted in a thermal regime that was generally not conducive to supporting juvenile salmon throughout the summer months. In contrast, the upper segment of the Mirror Lake site drained streams through mature riparian vegetation thought to provide thermal refuge during warm summer months. Streams draining from the upper portion of the study area maintained water temperatures deemed functional for supporting juvenile salmon (Jones et al. 2009).

Water-temperature (max temperature 25-30°C; Sagar et al. 2011) appeared to limit the opportunity for juvenile salmon to benefit from the lower portions of this site during summer months (see temperature section 3.2.1.2). In addition to affecting juvenile salmon access to the site, the thermal regime may also explain characteristics of fish community composition at the Mirror Lake sites. Non-native species composed half of the fish species composition at the lower sites (i.e., warmer regions), yet, only one non-native taxon was collected from the creek sites in the upper portion (i.e., cooler region) of the study area (Sagar et al. 2011). In addition to water temperature, it is likely that elements of habitat complexity and connectivity as described above may explain the variation in fish community composition at Mirror Lake.

Juvenile Chinook salmon were typically present at the sites closest to the main-stem of the Columbia River and were generally present throughout the study period (April through August). Chum salmon were only captured during May and their occurrence was also limited to sites near the main-stem. Juvenile coho salmon dominated catches at the cooler, upstream stream sites and, depending on the

specific location of sampling, their occurrence persisted through spring and summer months (Sagar et al. 2011). Efforts aimed at collecting diet and prey data relevant to juvenile salmon were completed at the Mirror Lake sites, but the results were not available at the time this review was conducted (Jones et al. 2009; Sagar et al. 2011).

To evaluate the performance of juvenile Chinook salmon at the restoration site, the lipid contents and growth rates of juvenile Chinook salmon were examined. These attributes did not differ across the study area, nor did they differ from nearby study areas within reach H. Contaminant concentrations (DDTs, PCBs, and PBDEs) in fish tissue were generally similar between years at the lake sites; however, compared to nearby sampling areas in reach H, the concentrations of contaminants in juvenile Chinook salmon were higher at the Mirror Lake study area although relatively low in comparison to urban areas such as Portland and Vancouver (Sagar et al. 2011). The researchers indicate "as yet, we have seen no clear trends in salmon catch per unit effort, condition factor, lipid content, or growth rate that can be attributed to the habitat improvements made during the past years" (Sagar et al. 2011). It is unclear whether a formal analysis aimed at examining the Mirror Lake restoration actions on the performance of juvenile salmon will be forthcoming. In the absence of pre-restoration monitoring data and a clearly defined reference site, it may be challenging to conduct such an analysis.

5.10 Conclusions

Several positive trends were observed in the reviewed studies. At Crims, Kandoll Farm, and Ft. Columbia, hydrologic reconnections increased the opportunity for fish to access restored sites. In terms of evaluating capacity, improvements in water temperature were noted at Kandoll Farm and South Slough and improvements in prey production were noted at Crims Island. A positive benefit of realized function was observed at Crims Island by examining residence time. Based on the available AE research findings, restoration in the LCRE appears to offer positive benefits to juvenile salmon in terms of opportunity, capacity, and realized function.

AE data limitations hampered the ability to draw conclusions regarding juvenile salmon benefits associated with habitat restoration. Of the 42 aquatic restoration projects completed in the LCRE since 2004, only a small fraction (n=9) included AE monitoring that addressed elements relevant to juvenile salmon ecology (i.e., opportunity, capacity, and realized function). In many cases, AE research lacked pre-restoration data, reference sites, and/or statistical analyses aimed at specifically evaluating response of monitored metrics within the context of restoration actions. Of the existing nine AE projects, most were conducted in the lower 90 rkm of the estuary, six included reference sites and pre-restoration monitoring, and one included a formal statistical analysis to evaluate response of metrics from restoration actions. This situation presents significant challenges with respect to effectively evaluating salmon performance in restored sites and across the landscape.

AE monitoring often includes a variety of metrics not directly related to salmon performance. These metrics often involve physical and structural conditions (e.g., vegetation, channel cross section, elevation). While some of these metrics can be used to infer the opportunity for juvenile salmon to access sites (e.g., temperature, extent of channel inundation) and the capacity of the sites to support juvenile salmon, there are limits to the inferences that can be made with respect to the functional response of juvenile salmon as a result of restoration. Of the nine projects reviewed, functional metrics were the least applied in evaluating restoration projects thereby constraining conclusions that can be made with respect to salmon performance as a response to restoration.

6.0 Status of Estuarine Ecosystem

In the previous sections, the status of knowledge about salmon habitat use and the factors believed to be limiting to salmon growth and survival in the estuary were summarized. In addition, the status of knowledge of restoration projects within the context of producing positive effects on salmonids were evaluated. The estuary ecosystem plays a vital role, but some elements of the ecosystem could be limiting in terms of supporting salmonids. Further, considerable uncertainty remains associated with the quantitative contribution of specific estuarine habitats to salmon. In this section, the question of whether overall estuarine conditions are improving, declining, or not changing in response to actions taken to improve the ecosystem is evaluated. To address this question we summarize what is known about major changes in the last ~20 years that would affect how we interpret scientific findings derived from the preceding sections. We summarize results that can be most directly applied to the CEERP for making decisions about what actions may be taken to improve estuarine conditions. These summaries have been designed to be updated in future synthesis memoranda.

Estuarine conditions are defined here as measureable attributes of the ecosystem. These attributes include floodplain wetland habitat area and composition, the connectivity among habitats, non-native aquatic species, water properties, and the sources and extent of stressors on the ecosystem and these attributes. We summarize recent work on factors that drive the spatial and temporal variations in many of the attributes as a way to inform future actions. Finally, we summarize information about how climate change may affect the condition of the estuary and restoration project planning.

The relevant syntheses that cover many of the topics addressed here include Small et al. (1990), which presents the results of detailed studies conducted as part of the Columbia River Estuary Data Development Program (CREDDP); the paper by G. Johnson et al. (2003), which synthesizes information about changes in the estuary relative to development of an ecosystem-based approach to habitat restoration; and the report by Bottom et al. (2011), which summarizes six years of research on salmonid biology and ecology in the estuary. The CREDDP, conducted in the late 1970s through the early 1980s, represents the "…first integrated, process-oriented series of studies with an ecosystem perspective" in the estuary (page 4, Simenstad et al. 1990). The report by Simenstad et al. essentially provides the most comprehensive benchmark in time for comparison here.

As part of the CREDDP studies, Sherwood et al. (1990) summarized historical changes in the Columbia River estuary. They found large changes in estuarine morphology caused by navigational improvement and by diking and filling of much of the wetland area. The tidal prism had decreased by 15% and there had been a net accumulation of sediment in the lower estuary. River flow had been significantly altered by regulation and diversion of irrigation water. Flow variability had been dampened, and net discharge had slightly reduced. Mixing had been reduced, stratification had increased, and salinity intrusion length and transport of salt into the estuary had increased. Sherwood et al. (1990) calculated an approximate reduction of 85% in wetland plant production, a 15% reduction in algal production, and a combined reduction of ~52,000 MT/yr of organic carbon input to the estuary. This loss had been partially supplanted by an increase in fluvial plankton produced in the reservoirs. The net result has been a major change in the organic matter sources supporting the estuarine food web. Sherwood et al. (1990) concluded that these major modifications need to be incorporated into contemporary estuarine and shorelands management especially those that would further exacerbate the altered conditions.

6.1 Methods

The general null hypothesis is that there is no detectable change associated with actions taken to improve the LCRE ecosystem. Below, we define the methods and metrics used to evaluate change.

6.1.1 Approach Used to Evaluate Change

We based our conclusions regarding the direction of change on the preponderance of results from a variety of sources. We chose this approach because there is no integrated, long-term, and comprehensive monitoring of the selected ecosystem attributes. Hence, we have to rely on incomplete data and indirect sets of information to develop inferences about the system condition. Essentially, this is designed to objectively "roll up" disparate data sets and information to formulate and support an objective, accurate, and repeatable assessment.

6.1.2 Metrics Used to Assess Change

The attributes we considered for assessing the trends in the status of the ecosystem are listed in Table 6.1. The metrics used to indicate the status of the attribute are also listed. These are based on a review of the literature, as well as recommendations derived from recent research in the LCRE. Changes were determined by our interpretation of inference based on direct or indirect data collected in the system.

Table 6.1. The Attributes, Metrics, and Conditions of the Attributes Considered for Evaluating Changes in the Columbia River Estuary Ecosystem, and a Summary of Conditions at Three Points in Time

Attribute	Metrics	Historical or Baseline Condition	CREDDP Condition Description (1978–1984; Small et al. 1990)	Relative Change Since CREDDP (1985-present)
Floodplain Wetlands	Area, species composition, similarity to references, connectivity	Minimal diking/filling/ conversion	Majority lost/altered	Restoration projects on various scales and in various habitats; vegetation more similar to reference sites; watersheds continue to be altered
Hydrology	Flow rates, timing and duration of freshet	Unrestricted flows from watersheds	Highly regulated	Highly regulated
Water Properties	Temperature, salinity, dissolved oxygen, organic matter, clarity, chlorophyll a, current velocity	Minimal alteration	Higher water temperatures; altered salinity regime	Higher water temperatures; altered salinity regime

Table 6.1. (contd)

Attribute	Metrics	Historical or Baseline Condition	CREDDP Condition Description (1978–1984; Small et al. 1990)	Relative Change Since CREDDP (1985–present)
Food Web	Organic matter source contribution, salmonid prey species	Dominated by floodplain wetland organic matter and estuarine-produced prey species	Dominated by production in reservoirs	Dominated by production in reservoirs; evidence that restoration sites are contributing marsh macrodetritus
Benthos	Dominant taxa		Altered by dredging; sediment contamination evident	Altered by dredging; sediment contamination evident
Ecosystem Processes	Wetland accretion, marsh macrodetritus export	Sediment and organic matter exchange and disturbance regime minimally altered	Limited because of restricted connections and loss of habitat	Limited because of restricted connections and loss of habitat. Some improvement because of restoration projects
Biodiversity	Non-native wetland species, non-native fish species, non-native zooplankton species	Native	Moderately to highly affected by non-native invasive species	Moderately to highly affected by non-native invasive species. Invasions at high rate over past few decades
Stressor Level (human-caused)	Additive site-based stressor score	None	Moderate	Moderate

6.2 Analysis of Attributes

Our approach centered on evaluating data relevant to both ecosystem processes and structural attributes. Ecosystem processes include factors that form and maintain habitats and water properties, as well as ecological functions (e.g., organic matter export). Structural attributes encompass commonly measured elements of the ecosystem that indicate status and for which there is a strong scientific underpinning. All of these attributes have changed because of human interventions of various kinds, and would be expected to improve with restorative actions. Taken together, they cover the suite of elements that are recognized globally as important aspects of aquatic ecosystems.

Although stressors are typically not an ecosystem attribute, we have included them here for evaluation purposes. Stressor scores have been developed for approximately 2100 hydrologically defined sites in the historical LCRE floodplain (Evans et al. 2006; Thom et al. 2011b). Theoretically, restoration actions should reduce the stress score at sites and cumulatively in the ecosystem. In turn, the greater the reduction in stress score, the greater the improvement in ecosystem condition.

6.2.1 Floodplain Wetlands

6.2.1.1 Area

A key attribute of the estuary condition is the size, location, and condition of the benthic habitats, especially those that comprise floodplain wetlands, unvegetated flats, and channels. Fundamental elements and processes have been altered in the estuary beginning in the 1800s. The estuary has undergone large changes in bathymetry and topography caused by navigation improvement projects such as jetties, pile dikes, and dredged channels (Sherwood et al. 1990). Diking and filling have resulted in loss of much of the wetted area in the floodplain. Changes in hydrology have resulted in displacement of sediment and various realignments of the mouth of the estuary. Because of the importance of these habitats to fish as well as other resources in the system, concentrated focus has been placed on restoring floodplain habitats.

Thomas (1983) documented large-scale losses of tidal wetlands in the lower portion of the estuary since the 1800s. The LCREP (2012) produced a system-wide assessment of habitat changes from Bonneville Dam to the mouth. Overall there has been a loss of 118,971 acres of tidal wetland habitat. Among the major vegetation classes, forests lost 56,565, wooded lost 33,688, and herbaceous classes lost 28,718 acres (Table 6.2). A river reach-by-reach analysis (Figure 6.1) shows the following:

- Conversion of wetlands to agricultural land was extensive, especially in reaches A (rkm 4−23), B (rkm 23−61), E (rkm 119−137), and F (rkm 137−165).
- Conversion to developed property was extensive in reaches A, D, F, G (rkm 165−204) and H (rkm 204−233).
- Conversion and loss of forested wetland was pronounced in all reaches except H.
- Conversion and loss of herbaceous wetlands was extensive in all reaches except H, where the floodplain was naturally very limited because of the narrow geometry of the reach.
- Expansion of tidal flats occurred in reach B (attributed to reduced flow rates and deposition, Sherwood et al. 1990).

Wetland habitat restoration and protection started in the 1990s. Since 2001, approximately 46 confirmed aquatic (e.g., hydraulic reconnections, channel creation, LWD placement) restoration projects have occurred in the LCRE. According to the latest information from LCREP, these actions have restored a total of 2991 acres. Acquisition and non-aquatic based restoration (e.g., re-vegetation, invasive species control) over this same period has led to the protection/conservation of 7972 acres (K. Marcoe, LCREP, personal communication, 3 July 2012). About 376 acres were added between 2010 (Table 6.2) and 2011 (Figure 6.2). The projects are distributed throughout the estuary, with notable concentrations in reaches A and B (Figure 6.2).

Estimating habitat area changes is complicated by the uncertainties associated with interpreting habitat types from historical bathymetry and topography surveys, using imagery for classification of habitat types, and resolving changes from projects that cover relatively small areas or are very new. An additional consideration is the baseline from which to assess future changes. A possible solution is to develop a database for all projects that includes information about the location, size, and type of habitat restored or protected. The LCREP data set provides a very good start for this. Settling on a standard habitat classification system, presently being developed (Simenstad et al. 2011), should help minimize the issue of variation in habitat type designations.

Table 6.2. Analysis of Floodplain Habitat Changes (gains, conversions, and losses) Between the Late 1880s and 2010 (LCREP 2012). TWL = tidal wetland; W = wooded; WL = wetland.

Forested Class	Total Area of Change (acres)
Forest to Herb Tidal WL	318.8
Forest to Wooded Tidal WL	730.3
Gained Forest	6878.0
Intact Forest	25,354.8
Lost Forest	56,564.8
Tidal WL to Forest	1055.9
Unclassified to Forest	360.4
Wooded Classes	**Total Area of Change (acres)**
Changed TW: Herb to Wood	1968.6
Changed TW: W to Herb	1306.2
Gained Wooded TWL	2369.4
Intact Wood TW	4184.5
Lost Wooded TWL	33,678.7
Tidal Flat to WTWL	432.1
Wooded TWL to Tidal Flats	270.0
Herbaceous Classes	**Total Area of Change (acres)**
Change Type: Wood to Herbaceous	1306.2
Changed Type: Herb to Wooded	1968.6
Grained herb Tidal WL	4124.5
Herb WL to Tidal Flat	902.2
Intact Herb Tidal WL	3876.7
Lost Herb Tidal WL	28,718.3
Tidal Flat to Herb TWL	1326.4

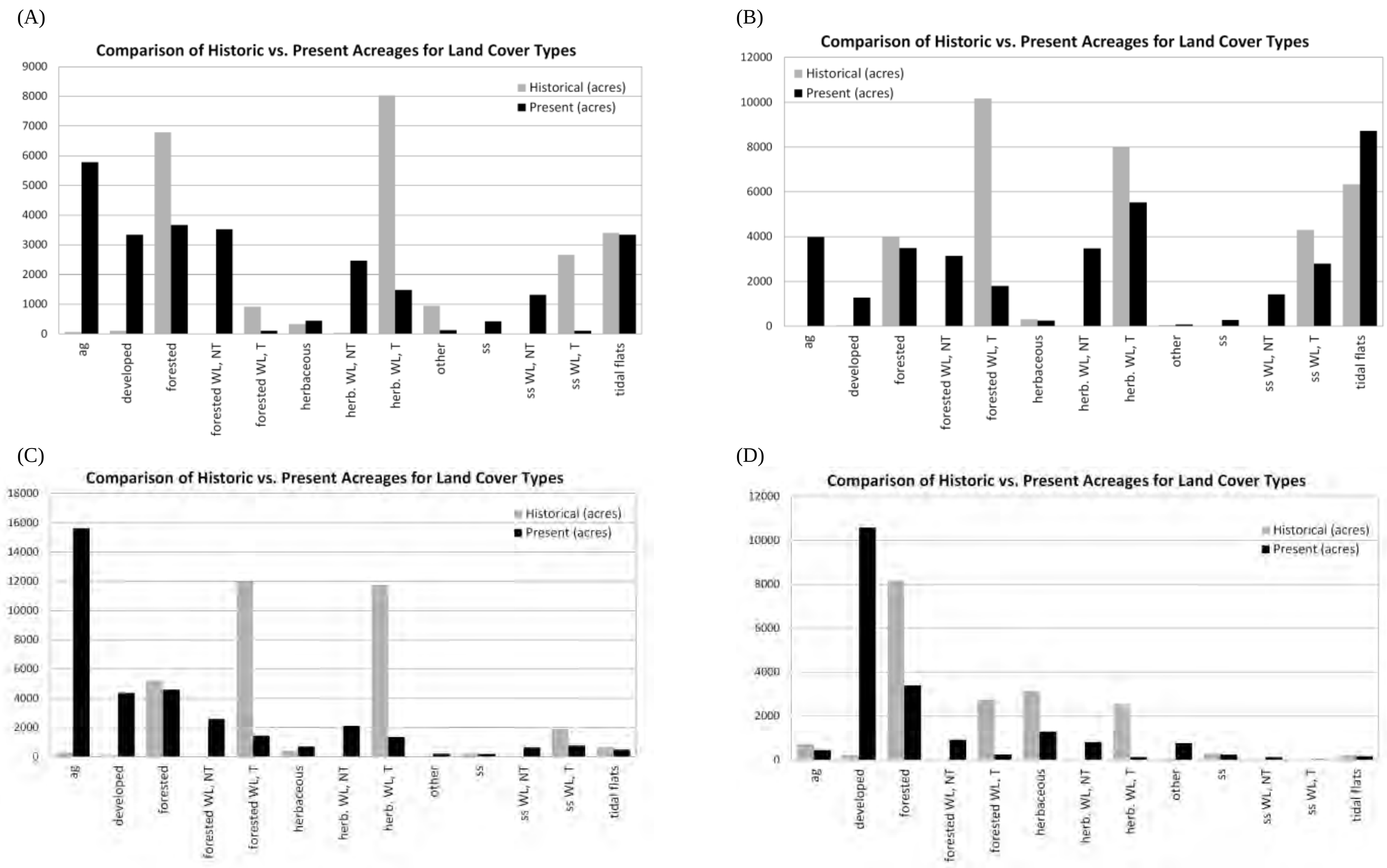


Figure 6.1. Comparison of Floodplain Wetland Habitat Changes in the Eight Reaches (A−H) Between the 1880s and 2010 (LCREP 2012).

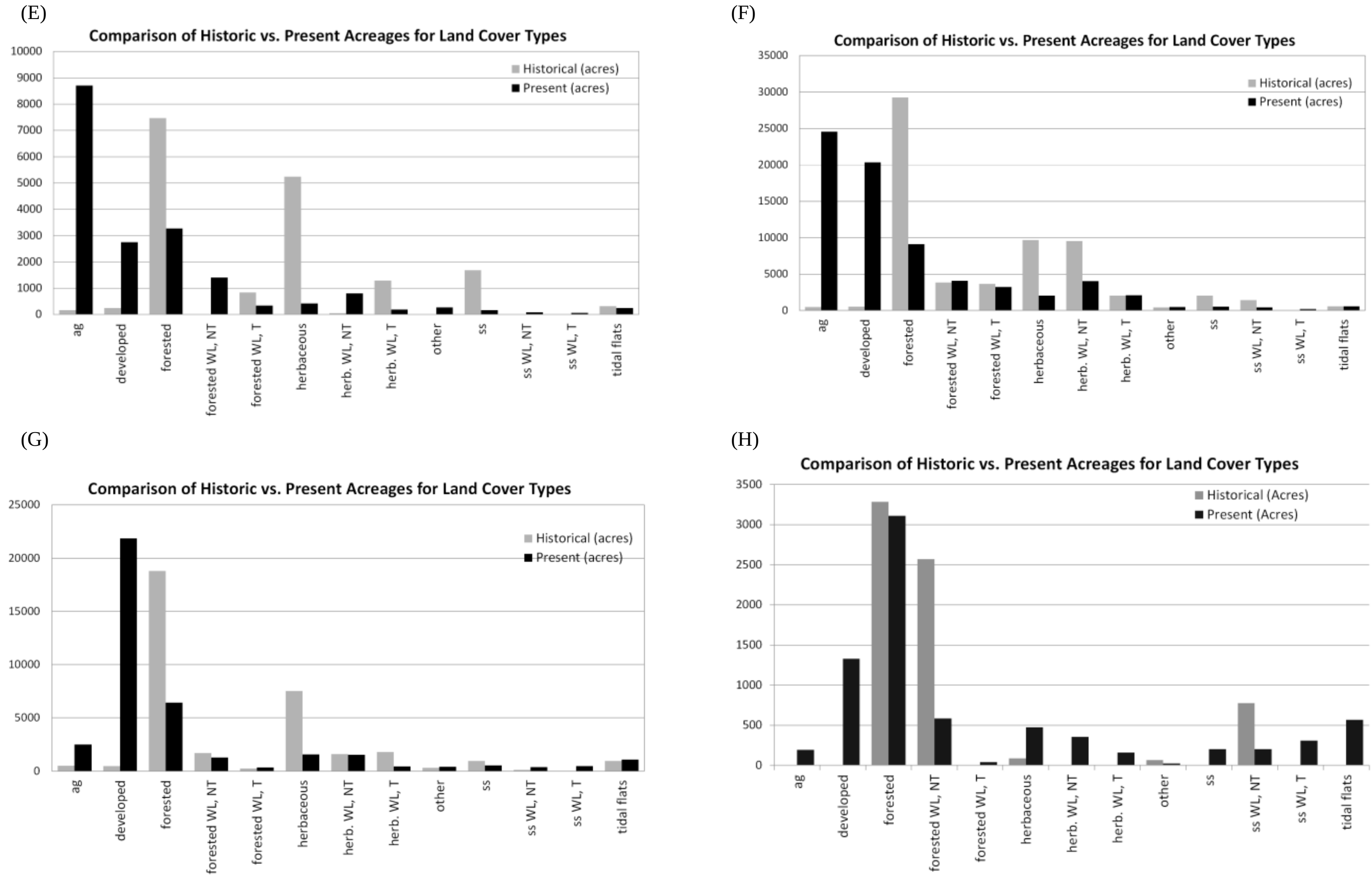


Figure 6.1. (contd)

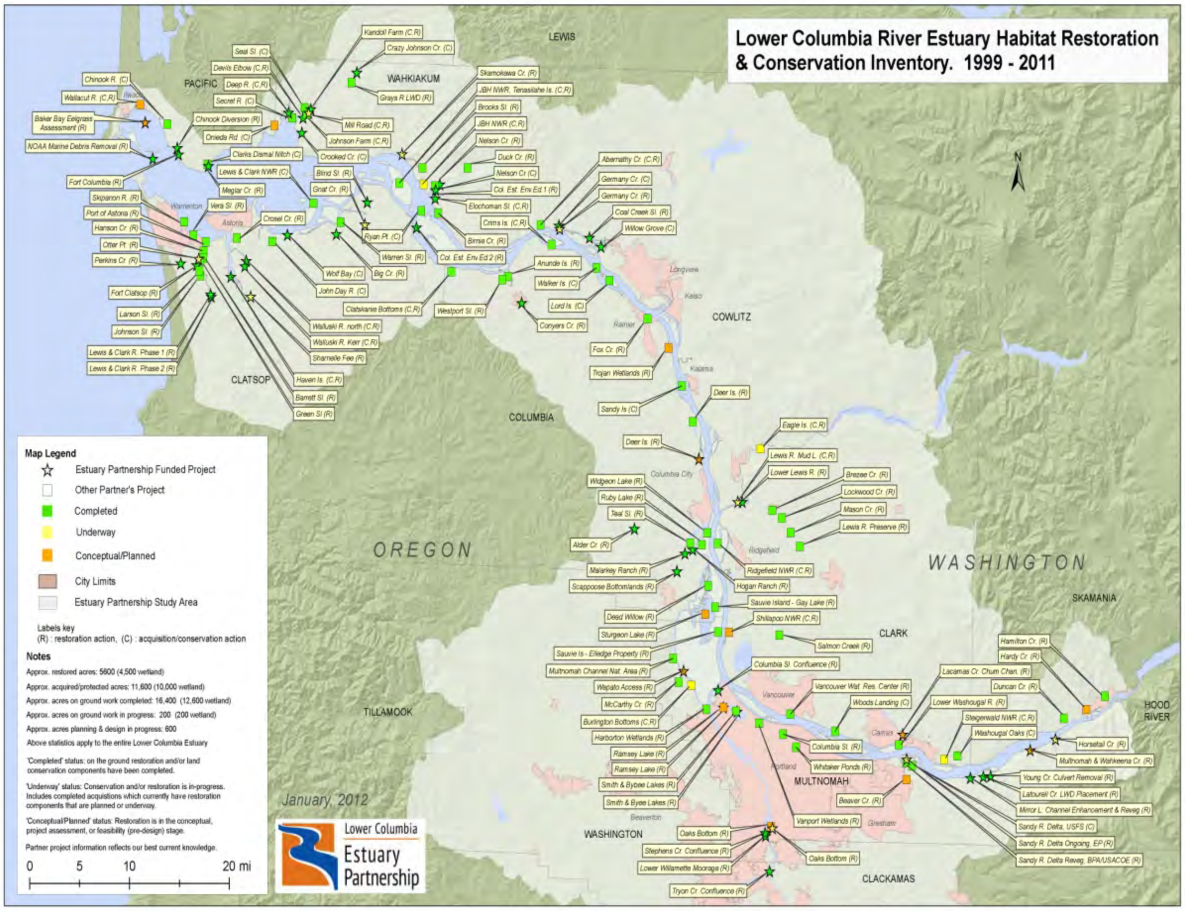


Figure 6.2. Map Showing Restoration and Conservation Projects Completed, in Progress, and Planned in the Columbia River Estuary. (Photo courtesy of K. Marcoe at the Lower Columbia River Estuary Partnership)

6.2.1.2 Vegetative Species Composition

Here we address the question of whether the plant species composition in floodplain wetlands is changing. Although there is no comprehensive analysis of species change over time, we can partially address the question through analysis of reference sites and restored or restoring sites. The two metrics of change we used are 1) whether the reference sites species composition has changed because of either disturbances or invasive non-native species, and 2) whether the vegetation species assemblage at restored sites is approaching that of reference sites.

The most comprehensive analysis that has been done on the topic of species composition is associated with the reference site and habitat monitoring studies conducted through the LCREP (Sager et al. 2011; Borde et al. 2012). That work sampled vegetation species cover at 51 reference sites, 29 monitoring sites, and 10 restoring sites (i.e., restored, previously breached, created; Figure 6.3). A total of 280 species of wetland-associated macrophyte species were recorded over all sites.

In developing reference sites in emergent wetland habitats in the estuary, Borde et al. (2012) evaluated the effects of the origin of the site (i.e., natural or created) and hydrology. In general, natural

sites contained more species and more below-ground biomass than created sites. Among the most striking results was the strong relationship between hydrodynamics and species richness. Sites close to the river mouth and close to the dam contained the lowest number of species, and sites located between these two extremes contained the greatest number of species. Vegetative distribution is driven by stresses from salinity and the relative large tidal variation near the mouth as contrasted with extended periods of very high water near the dam. Further, there was a strong negative correlation between water level and percentage cover of three dominant species: common spike rush (*Eleocharis palustris*), wapato (*Sagittaria latifolia*), invasive reed canarygrass (*Phalaris arundinacea*). Cover was greatest during low-water years (e.g., 2005), and lower during high-water years (e.g., 2011). Finally, Borde et al. (2012) believe there are very few truly “historical emergent marshes” in the estuary based on their analysis of historical maps from the 1800s and the present location of these features. This suggests that these marshes naturally “migrated,” probably because of natural forcing from the river and tides, sediment accretion and progradation, and perhaps human alteration of the morphology of the estuary. Changes in these forcings from historical conditions may constrain contemporary evolution of wetland systems. These spatial and temporal patterns, and what drives them, are critical to understand when planning and evaluating the restoration projects.

Based on an analysis of vegetation assemblage composition in emergent marshes, and water level dynamics, Borde et al. (2012; and personal communication, May 2012) divided the estuary into five emergent marsh (EM) zones (Figure 6.3). Borde et al. (2012) used several different lines of evidence based on vegetation species richness, species composition, salinity, and inundation to determine vegetation distribution patterns along the estuarine gradient. Below rkm 29, salinity is a factor affecting the lower estuary vegetation distribution patterns. Fluvial flows are a factor between rkm 29−104, but inundation is predominantly tidally driven; the amount of inundation that occurs at all elevations during the entire year is equal to or greater than the amount that occurs during the growing season. Above rkm 104, there is a shift in the timing of inundation: a greater proportion of the inundation occurs during the growing season, and is not spread throughout the year. However, the magnitude and duration of inundation is still low relative to the zones farther upriver. The magnitude and duration of inundation during the growing season begins to increase between rkm 136 and 181. The Borde et al.(2012) analysis of inundation at an elevation of 2 m (Columbia River Datum) showed that the slope of the $\log_{10}$ inundation value as a function of river kilometer is not significantly different from zero below rkm 136 ($p = 0.96$), whereas inundation during the growing season was considerably higher above this point. EM zone 5, the closest to the dam, is the most fluvial-dominated zone. Inundation is very high during the growing season when the spring freshet occurs, and it is very low during the rest of the year when flows are very low to moderate.

The boundaries of the observed vegetation zones are shown below. In evaluating the spatial boundaries of the hydro-vegetation zones, the following points need to be considered:

- The hydrologic part of the analysis was based on data from 37 floodplain wetlands.
- The boundaries are also based on 22 least-disturbed marshes used in a discriminant function analysis of cover data for 13 plant species between rkm 12 and 230.
- The boundary between zones 4 and 5 is a rough estimate due to the lack of sites in this area.
- The hydrologic data were collected during a limited period, between 2008 and 2010, and different results may be observed using data from different years.

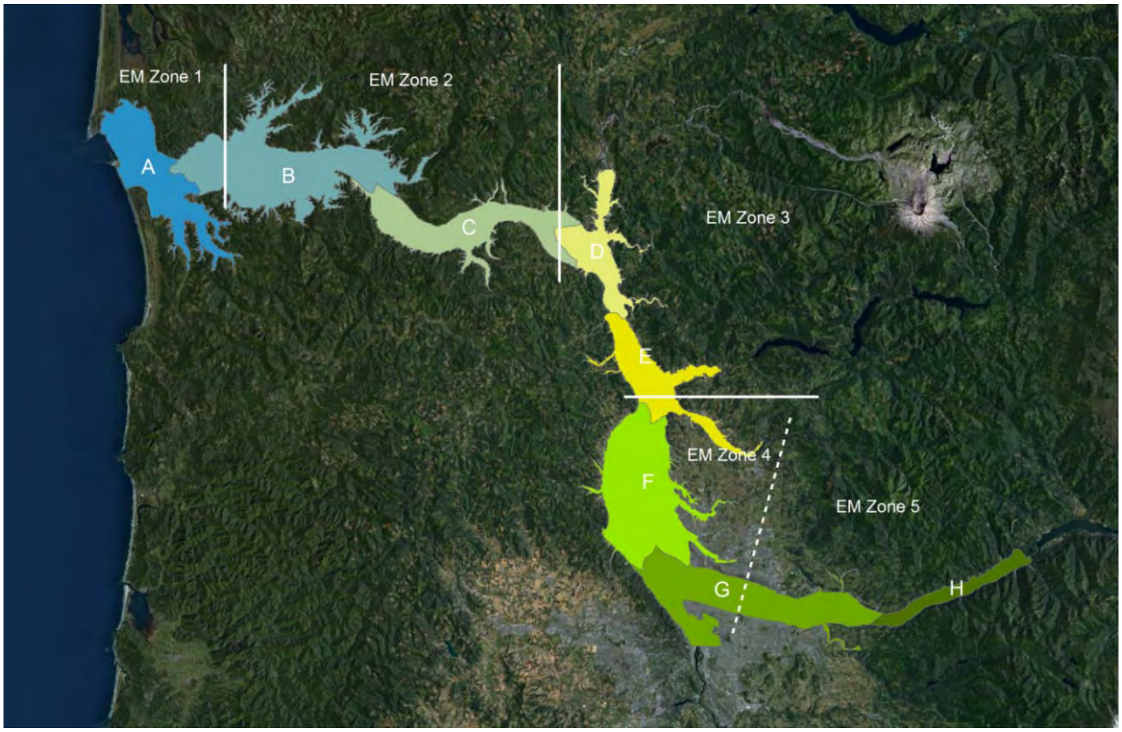


Figure 6.3. Hydrogeomorphic Reaches (A−H; Simenstad et al. 2011) and Emergent Marsh Zones (EM zones 1−5; Borde et al. 2012) of the LCRE. Solid lines indicate boundaries of the emergent marsh zones. The single dashed line indicates boundary delineation is tentative until additional data can be used to confirm the location of this boundary.

The link between hydrodynamics and vegetation is important to understand in order to design and predict the outcomes of restoration projects, understand interannual variation in the assemblages, and predict the effects of larger-scale changes such as climate changes.

6.2.2 Hydrology, Water Properties, and Food Web

Major changes occurring in the riverine and estuarine water quality and food web since the installation of the dams along the main-stem of the Columbia River have been well documented. Dam construction and subsequent channel diversions, irrigation activities, and dredging have altered the river flows in terms of their timing and magnitude, resulting in a decreased overall river discharge and dampened seasonal flow variability (Sherwood et al. 1990). The peak flow of the Columbia River occurs in the late spring during the freshet, and the lowest flow occurs in late summer to early autumn. The presence of the dams has shifted the system from a high-turbidity, detritus-driven river ecosystem to a much “greener” river, where pelagic primary production (i.e., fluvial phytoplankton) has increased as a result of the reduced load and longer water residence time behind the dams (Sullivan 2001). In fact, the water in the Columbia River today has relatively low turbidity (10−30 mg/L) (Sullivan et al. 2001; Prahl et al. 1997, 1998) compared to some of the world’s major rivers. An exception is the estuarine turbidity maximum (ETM), which can show a three orders of magnitude increase at times (Crump et al. 1988). Along with altering the physical processes, the change in flow has affected the dynamics of the ETM zone (Simenstad et al. 1994). The ETM zone is an important zone of concentration of planktonic organisms, particulates, and dissolved matter, and where recycling processes and deposition to the bottom can be intense.

The basic water-quality conditions relevant to salmon health (e.g., temperature and DO concentration), and water properties associated with food web resource assessment and habitat (e.g., primary productivity, chl *a*, nutrients, photosynthetically active radiation (PAR), plankton composition, organic matter characterization) have been altered extensively since historical conditions. With an estimated >80% reduction in emergent plant production, and a commensurate increase in fluvial phytoplankton in upriver reservoirs, the primary productivity regime has shifted from one based on emergent vegetation to one dominated by pelagic, or fluvial, phytoplankton (Simenstad et al. 1990b; Sherwood et al. 1990; Small et al. 1990; Bottom et al. 2011).

To support characterization of salmon habitat capacity, the U.S. Geological Survey (USGS) has conducted seasonal water-quality monitoring at four fixed sites for conditions relevant to salmon health (e.g., temperature, and DO concentration; Sagar et al. 2012). The sites include Ilwaco (reach A, rkm 6; monitored in 2011), Whites Island (reach C, rkm 72; monitored in 2009 and 2011), Campbell Slough (reach F, rkm 149; monitored for 4 consecutive years), and Franz Lake (reach H, rkm 221; monitored in 2011). These sites were selected as representative off-channel salmon habitats. All sites showed daily and seasonal variation of basic water-quality parameters (temperature, DO concentration, pH, specific conductance), as well as inter-site variability based on overall site conditions and locations of the sensors. In addition, in 2011, the year all sites were monitored, each site experienced periods of "poor" water quality with respect to conditions for salmonid health during the spring and summer, although the duration of poor water-quality periods varied among sites. Poor water quality was defined as warm water (water temperature >17.5°C), low DO concentrations (<8 mg/L), and high pH (>8.5) (Sagar et al. 2012).

Additional efforts at these four sites in 2010 and 2011 have focused on developing and testing methods for assessing additional water-property conditions relevant to food web resource assessment (nutrients, PAR, algal biomass and species, algal productivity, stable isotope ratios) (Sagar et al. 2012). In most cases, these parameters were sampled three to six times between April and June at each of the four fixed sites. For the 2 years sampled (2010 and 2011) at Campbell Slough, there was no clear trend in total nitrogen concentration between sampling periods or between years (2010 and 2011). However in 2011, total nitrogen and total phosphorus were highest in June at all sites, except Whites Island where concentrations peaked in May. Organic nitrogen constituted most of the total nitrogen at all sites. The available nitrate was reduced greatly by late June, presumably taken up by organisms or flushed out of the system. Phytoplankton biomass, measured as chl_*a*, was generally between 5−10 mg/L between May and July of 2011 at all sites, with the exception of the Ilwaco Harbor where concentrations were >150 mg/L in April and became undetectable later in the season (Sagar et al. 2012). These high concentrations are likely due to the mixotrophic ciliate *Myrionecta rubra* (Herfort et al. 2011). Other studies have shown chlorophyll concentrations to be greatest during the spring diatom bloom; reductions occurred in the summer and minimal values occurred during the winter (Prahl et al. 1998; Sullivan et al. 2001; Roegner et al. 2011b).

Additional water-column components of the lower food web (phytoplankton and zooplankton abundance and composition) were examined at these four sites between early April and late May in 2011 (Sagar et al. 2012) as part of the habitat capacity assessment. Diatoms dominated the phytoplankton assemblage during the early spring when abundances were highest, as has been shown in other recent studies as well. Comparative samples from the Land-Ocean Biogeochemical Observatory (LOBO) monitoring site at the Beaver Army Terminal (BAT) also indicated multiple peaks of chl_*a* during the spring and summer periods. These high-resolution in situ time-series have revised the understanding of fluvial phytoplankton dynamics during the spring bloom period from prior studies based on monthly or

weekly samples (Sullivan et al. 2001; Roegner et al. 2011b). River discharge volumes were inversely related to chl_*a*. A recent comparison of observations from the shallow-water habitat sites to the main channel at BAT indicate that phytoplankton abundances can be 10 times higher in the shallow-water habitats, as long as river discharge is low relative to the freshet.

Recent isotopic evidence (Bottom et al. 2008; Maier and Simenstad 2009) has suggested that salmon prey preferentially consume organic matter derived from vascular plants (emergent vegetation detritus), although the organic matter driving estuarine production is currently derived primarily from fluvial phytoplankton (approx. 58%; Small et al. 1990).

The Cumulative Effects (CE) study (G. Johnson et al. 2012) has shown through a modeling effort at Kandoll Farm and the mouth of the Grays River (rkm 37) that a substantial proportion of organic matter (marsh macrodetritus) can be exported to the main-stem of the Columbia River. Approximately 52% of the material exported would reach Grays Bay, and 48% would remain in the floodplain, available for incorporation into the food web.

6.2.2.1 Recent Status of Water-Quality Conditions

It is difficult to assess the status of recent water-quality conditions with respect to improvements or declines, primarily because of the lack of continuity of long-term data sets necessary to understand trends, particularly with respect to tidally influenced emergent wetlands. The monitoring at the four fixed estuary monitoring program sites is designed to provide information about these trends, but 2011 was the first year during which water quality was monitored and primary productivity was assessed at three of the four fixed sites. Thus, the LCREP report (Sagar et al. 2012) has been able to provide a 1-year status assessment at those sites. Not surprisingly, the multiyear data set from Campbell Slough shows annual variability in water-quality parameters during years with different hydrologic conditions and weather patterns (Sagar et al. 2012) which further demonstrates the need for long-term data in order to understand water-quality trends.

The type and magnitude of recent restoration actions may greatly influence the magnitude of realized change that can occur with a variety of water-quality parameters. For example, temperature changes that occurred at three primary restoration projects and associated reference sites (Crims Island, Kandoll Farm, and Vera Slough) as part of the CE study varied depending on the habitat type and the type of restoration action undertaken (e.g., tide-gate replacement, culvert replacement, dike breach). Monitoring at those sites indicated that the greater the extent of the hydrologic reconnection, the greater the return to temperatures in adjacent open waters (G. Johnson et al. 2012).

The higher abundances of phytoplankton noted in the shallow-water habitats relative to the main-stem river (Sagar et al 2012) suggest the availability of pelagic phytoplankton may be greater within protected areas. This greater availability highlights the importance of emergent wetlands—not only for providing a source of vascular plant macrodetritus, but also plankton species, which dominate during the spring and early summer and are a source of nutrition for juvenile salmon (Maier and Simenstad 2009). Increasing the physical complexity of habitats (added wetlands and channels) may also serve to trap organic matter from fluvial phytoplankton sources, thereby providing additional nutrition for salmon prey (Sagar et al. 2012).

6.2.3 Benthos

Within the 20-year period from 1988 to 2008, at least eight studies of the main channel habitats in LCRE included sampling and analysis of infaunal invertebrates within the overall study designs. Geographically the studies spanned the area of the river from near its mouth to Bonneville Dam. All of the studies were short-term (<3-year) events that were conducted to answer specific research questions, not to build a comprehensive picture of infaunal communities throughout the estuary. Studies in the late 1980s and the 1990s primarily focused on evaluating benthic habitats to answer questions about habitat restoration, dredged material disposal, or food availability for white sturgeon. Studies in the 2000s focused mainly on salmon food availability and habitat restoration. Partly because of serving different purposes, the studies used several sampling approaches and methodologies that make comparisons among them and building a big picture view somewhat tenuous. For example, several studies used relatively small core samplers to collect infauna, and the samplers differed in the area sampled from about 11.6 to 24 cm^2. Other studies used more standard 0.1- m^2 grab samplers to collect infauna. Nonetheless, despite the temporal and methodological differences, some patterns do emerge from this patchwork of data that provide information about infaunal communities within the estuary that could serve to form hypotheses for further studies.

Probably the most striking pattern that emerges is that infaunal densities within the estuary vary tremendously on small and large geographic and temporal scales. This variability is often large enough that it overwhelms the ability of statistical testing to detect significant differences. For example, McCabe and Hinton (1996) sampled beach nourishment sites 30 m riverward of the high-tide marks between rkm 53 and 122 and reported no statistical difference in infaunal densities among the 4 months sampled in 1994 and 1995. However, the lack of statistical significance was most certainly because of extreme variability among individual samples, pooled samples (mean), across months and locations. For example, at Area W-43.8, infaunal densities did not differ statistically between months despite ranging from 3056 organisms/m^2 (July 1994) to 27,273 organisms/m^2 (January 1995), about a nine-fold difference. Hinton et al. (1995) studied a potential dredged material site and nearby shallow subtidal site from rkm 40 to 42 and reported that variability within a relatively small area was large. For example, within a potential restoration area of about 0.55 km^2, mean infaunal densities in May 1993 ranged from 945 to 47,502 ind/m^2 and individual station variability was often high (coefficients of variation often were >35%). Within a smaller shallow subtidal area (0.36-km^2 area) densities varied from 7216 to 61,074 ind/m^2. The variability in habitats studied and sampling methods hamper confident evaluation of general density trends across the estuary, although densities in the upper estuary (beyond rkm 121) appear to be substantially less than elsewhere (<2000/m^2; McCabe et al. 1997).

Another noticeable pattern is that within the estuary, the predominant fauna composing the infaunal community are relatively consistent, albeit with some geographic and temporal variation. Corophiid amphipods, particularly *Americorophium* (formerly *Corophium*) *salmonis*, are among the predominant infaunal taxa in the community whether near the river mouth (Hinton and Emmett 2000), mid-estuary (Hinton et al. 1995; McCabe and Hinton 1996; Haskell and Tiffan 2011), or upriver from rkm 121 (McCabe et al. 1997). Other predominant taxa often include oligochaete and nemertean worms, chironomid (non-biting midge) larvae, ceratopogonid (biting midge) larvae, and introduced Asian clams (*Corbicula fluminea*). For example, *A. salmonis* accounted for about 31% of the total infaunal abundance off Trestle Bay (Hinton and Emmett 2000), about 60% between Miller Sands and Pillar Rock Island (Hinton et al. 1995), and 46 to 91% off Gull Island (Haskell and Tiffan 2011). Asian clams and ceratopogonid larvae consistently were the most abundant taxa across stations and months sampled in the

upper estuary (above rkm 121; McCabe et al. 1997). Farther downriver, the two taxa typically were less predominant but still ranked within the most abundant 5 to 10 taxa (e.g., Hinton et al. 1995; McCabe and Hinton 1996; Haskell and Tiffan 2011). Oligochaete worms, which are typically not identified to species, are relatively abundant from the mid-estuary toward the river mouth.

It is commonly recognized that the benthos is a vital part of any estuarine ecosystem, not only because of its role as an integrator of environmental stresses and conditions, but also because of its dietary importance for key ecological and protected species. Although the LCRE is a large, geographically complex ecosystem, understanding the dynamics of the benthos within the estuary would benefit from a systematic sampling program specifically designed to answer key questions about infaunal geographic and temporal variability. Understanding infaunal community dynamics requires the use of consistent methodological sampling applied in accordance with a spatially and temporally rigorous design. A useful example is provided by the Chesapeake Bay Monitoring Program, which examines temporal patterns via a fixed-station design and spatial patterns via a probability-based, stratified simple random design to locate sampling stations (Llansó et al. 2007). The use of this or a similar approach would be an effective way to answer key questions about the condition of benthic communities in the estuary, and ultimately to translate the answers derived from the questions into an understanding of the overall well-being of the ecosystem.

6.2.4 Biodiversity

With loss of habitats and other disturbances, ecosystems tend to lose species. In addition, non-native species, especially those that are considered highly invasive, can colonize and dominate disturbed natural habitat. Research has shown that biodiversity can be a strong regulator of ecosystem function (e.g., Tilman et al. 1997; Reich et al. 2001). A comprehensive list of species does not exist for the estuary. However, several studies have been conducted at sites throughout the estuary that provide a workable list that could be evaluated by sampling the same sites in the future. As far as we know, no one has published a comprehensive list of species for any taxonomic group that existed in the estuary historically. Further, there have been no comprehensive efforts to develop a recent list. However, indications of changes in biodiversity can be derived from studies that covered large areas in the estuary.

The LCREP web site contains a long list of species reported from the LCRE, but the source of this list is unclear. There is also a list of wetland and upland plant species for the estuary developed from surveys of several sites conducted by John Marshall and others; it is organized as a VEMA database (GeoMobile Innovations Vegetation Management Microsoft Access). The database can be accessed through the Northwest Habitat Institute (http://nwhi.org/index/). Borde et al. (2012) have identified wetland plant species over a total of 80 sites spread throughout the estuary. These sites included habitat monitoring, reference sites under study through LCREP since 2005, as well as research sites begun in 2005. Borde et al. identified and quantified plant species occurring in multiple quadrats within wetland strata. Species area curves indicated that this sampling method likely encountered the vast majority of the species in the strata. Borde et al. (2012) observed 172 taxa: 115 in created marshes and 139 in historic marshes. Seven taxa, including reed canarygrass, common spikerush, wapato, Lyngby sedge (*Carex lyngbyei*), Canada waterweed (*Elodea Canadensis*), false loosestrife (*Ludwigia palustris*), and slough sedge (*C. obnupta*), made up 68% of the cumulative cover. Reed canarygrass occurred in 52% of the quadrats and accounted for 28% of the cover at all monitoring sites.

Sytsma et al. (2004), Bersine et al. (2008), and Cordell et al. (2008) report that the LCRE contains numerous non-native species of vegetation, benthic invertebrate fauna, zooplankton, mammals, and fish. Compared to other large aquatic systems, the Columbia River contains a moderate level of invasive vascular plant species (19), annelid (11), mollusk (6), and crustacean species (14). Some of the more dominant non-native species include the reed canarygrass, common reed, New Zealand mud snail, the Asian clam, and four species of Asian copepods. Nutria has also been introduced. This species can cause severe damage to wetland plant communities. Reed canarygrass dominates many wetlands in the region, and is documented to colonize and dominate restored wetlands. Similarly, the copepods, Asian clam, some fish species, and nutria alter the food web and structure of native species assemblages and landscapes.

Introductions of fish species began in the late 1800s, and have increased steadily through the late 20th century. As of about 1980, 23 non-native species of fish were recorded. Introductions of invertebrate species have increased exponentially between about 1940 and the present. As of about 2000, 35 non-native invertebrate species have been recorded. In recent fisheries research studies non-native taxa have been captured from a variety of habitats throughout the LCRE. From 2002 through 2007, Bottom et al. (2011) captured a total of seven non-native taxa across marine, estuarine, and tidal freshwater sites. The highest catches of non-native fish were associated with tidal freshwater sites. From the cumulative catch, American shad (*Alosa sapidissima*) and banded killifish were the most numerous. Within the tidal freshwater reaches of the LCRE near the Sandy River delta (rkm 190−208), Sather et al. (2011) identified 18 non-native fish taxa, which composed approximately 25% of the total abundance of fish from 2007 through 2010. The ratios of non-native to native species by density varied by season and were approximately 0.9, 0.3, 0.14, and 0.45 in winter, spring, summer, and autumn, respectively. The size distribution of the non-native taxa captured indicated juvenile life stages were occupying the shallow-water habitats sampled. The exceptions were banded killifish, which were thought to complete their life cycle in the shallow-water areas, and large (>200-mm FL) smallmouth bass (*Micropterus dolomieu*) that were captured during summer. While there appears to be potential for predator-prey interactions in shallow-water habitats such as those between smallmouth bass and juvenile salmon, Sather et al. (2011) suggest a more thorough investigation is necessary and should consider sampling techniques, predator movements, and ontogenetic feeding variability.

A more recent documentation of a non-native species has been the Amur goby (*Rhinogobius brunneus*), which was first found in the East Fork of the Lewis River in 2004. Since that time this fish has been documented at Crims Island (rkm 90; Haskel and Tiffan 2010), at sites between the Cowlitz and Lewis rivers (rkm 109−141; Sather et al. 2011), and near the Sandy River delta (rkm 190−208; Sather et al. 2011). This species is native to eastern Asia and is thought to have been introduced to the Columbia River via ballast water or aquarium trade. The widespread documentation of this species had led to speculation that it has become established and is successfully breading in the LCRE, although the ecological implications are unknown (USGS 2012). Restoration projects appear to be adding species to sites, thereby expanding the distribution of native and non-native species. This has been accomplished by planting desirable species (e.g., riparian plantings) and opening access of species to sites through hydrological reconnections (e.g., levee breaches).

Species lists and species area curves have been used to document changes in plant species richness after wetland restoration in the region (e.g., Thom et al. 2003; Thom et al. 2012). In all cases, within 2 to 4 years after restoration the number of species and species area curves exceed pre-restoration values. As these systems mature, we expect the species richness to level off and perhaps decline slightly, as larger

(K-selected) species expand to outcompete early colonizing (r-selected) species to dominate sites (Simenstad and Thom 1996; Thom et al. 2002). The limiting factors for expanding species numbers and species distributions are the 1) availability of species in the regional species pool, and 2) the ability of these species to reach and colonize restored sites.

6.3 Net Ecosystem Change

Assessing the net change in the ecosystem involves a review of human-derived stressors, inferences that can be applied to restoration projects, and changes in landscape, and comparison of ecosystem conditions to those noted in CREDDP studies.

6.3.1 Stressors

Stressors, in the present context, are factors that alter the natural undisturbed condition of the ecosystem. Stressors are related to change brought about by human activity. The most comprehensive assessment of stressors in the Columbia River was developed as part of a process for prioritizing restoration projects in the system (Evans et al. 2006; Thom et al. 2011b). This study used available geographic information system layers that represented complete coverage of the estuary. The stressors identified by the analysis included the following:

- hydrosystem flow alteration
- contaminants
- waterways listed in Section 303-d of the federal Clean Water Act (i.e., waterways exhibiting impaired water-quality conditions)
- navigation channel dredging
- population
- flow restrictions
- facilities of interest (that receive permits and may have impacts from discharges, landfills, etc.)
- industrial development agriculture
- diking
- pile dikes
- minor and major overwater structures
- marinas
- marinas protected by breakwaters
- dredged material disposal sites
- industrial shoreline, shoreline change (i.e., altered morphology)
- shoreline armoring
- invasive species
- road length
- hydro-road intersections
- development
- forested area (i.e., lower forest area had a higher stress score)
- riparian (scored inversely as was forested area).

Figure 6.4 illustrates the level of disturbance or stress using a scoring process that rates all stressors. The site scale refers to stressors active in about 2100 specific sites (each on the order of 100 acres in area) in the historical floodplain. The MA (management area) scale refers to the condition of the watershed unit (in this case the Hydrologic Unit Code [HUC]-6) within which the sites occur. The greater the score, the more disturbed a site and watershed is. Sites and watersheds in the lower left corner are the least disturbed. Most sites in the system are moderately to highly disturbed or otherwise altered. Because these stressors affect the functioning of the ecosystem, tracking how stressors are reduced or increased

and the corresponding ecosystem response is a robust way to quantify net change in the ecosystem condition. The database with the stressors and their scores is available for reanalysis as restoration actions are implemented. At present, data are not available in a format to fully evaluate how stressors and related degree of stress have changed over the past two decades.

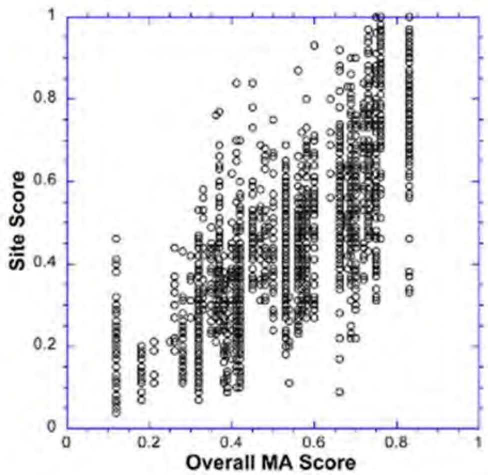


Figure 6.4. Site and Management Area (MA) Stressor Scores. These scores are derived from analysis of multiple stressors within approximately 2100 hydrologically defined sites and 60 watershed units (HUC-6; i.e., management areas) in the estuary. Higher scores indicate larger degrees of stress. (From Thom et al. 2011b)

6.3.2 Inferences from Restoration Projects

Although monitoring studies at restored sites are limited in number and scope (see Section 5.10), intensive monitoring at a number of restoration and reference sites in the estuary from 2005 through 2009 under the CE project (G. Johnson et al. 2012) revealed inferences that can be applied to restoration planning. These inferences, listed in Table 6.3, indicate that processes required to form and maintain floodplain habitat are generally restored once natural hydrodynamics are re-established at a site. They also illustrate that changes in the structure of the system occur rapidly during the 5-year post-restoration period (Thom et al. 2012, Chapter 2 in Johnson et al. 2012). Further detail regarding the source of information for these inferences can be found in the original report. A meta-analysis of AE monitoring data from restoring sites indicated that water temperature, sedimentation, vegetation structure, fish access, and the flux of organic matter produced were improved (G. Johnson et al. 2012). These improvements were evident after just 2 years.

Table 6.3. Summary of Findings and Inferences to the Site and Ecosystem Scales and Salmonids. This is Table 2.23 from Thom et al. (2012) which is Chapter 2 in Johnson et al. (2012). For further information regarding the source for these findings and inferences see the original document.

Finding	Inferences for the Site	Inferences for the Ecosystem	Inferences for Salmonids
Clear response of vegetation assemblage to restoration actions within 1 year following hydrological reconnection	Recovery of site habitat structure initiated quickly; restoration of natural biodiversity; enhanced site resilience	Processes associated with structure initiated within 1 year	Juvenile salmonid habitat access opportunity and feeding and rearing capacity are increased within 1 year
Initiation of sediment accretion	Will lead to restoring elevations lost through subsidence	Rapid vegetation assemblage development will extend for much longer than the 4 years of this study	Juvenile salmonid feeding and rearing capacity will change through time toward natural conditions
Redevelopment of historical tidal channels	Development of productive marsh edges and natural wetland morphology	Increased channel area and productive marsh edges in the floodplain; enhanced area for nutrient processing and export of organic matter	Juvenile salmonid habitat feeding and rearing capacity increased; enhanced organic matter export to estuarine ecosystem salmonid food web
Exposure of buried large wood and development of stepped pools in tidal channels	Development of natural wetland morphology to support microhabitat development, and natural biodiversity	Increased channel area in the floodplain; enhanced area for nutrient processing, organic matter deposition, secondary production	Enhanced quality for salmonid rearing and prey production in the floodplain
Improved water-quality conditions (e.g., temperature) where substantial hydrological connectivity was restored	Development of natural wetland water properties and support of aquatic species	Improved water properties in estuarine ecosystem	Enhanced quality for salmonid rearing and prey production in the floodplain and estuary
Frequent, prolonged, and repeated between-year use of restored sites by juvenile salmon	Natural biodiversity development	Natural ecosystem biodiversity development	Long-term enhancement of salmonid life history diversity
Use of tributary restored wetlands by “out of basin” fish	Natural biodiversity development	Natural ecosystem biodiversity development	Enhancement of salmonid populations and life history diversity in the ecosystem
Nutrient processing and organic carbon production	Development of natural wetland biogeochemical processes	Enhancement of natural ecosystem biogeochemical processes	Contribution of organic matter to support prey production in estuary
Export of marsh macrodetritus	Development of natural wetland primary production cycle	Enhancement of marsh macrodetritus entering the ecosystem; restoration of food web	Contribution of organic matter to support prey production in estuary

Table 6.3. (contd)

Finding	Inferences for the Site	Inferences for the Ecosystem	Inferences for Salmonids
Greater tidal reconnection produces quicker recovery	Quicker development of natural wetland structure and processes	Quicker recovery of ecosystem processes	Quicker recovery of support for salmonids
Evidence of potential synergism and optimization of projects	Site functions depend on sites surrounding them	Ecosystem functions depend on synergistic aspects of suites of sites	Synergistic support of salmonids through opportunity and capacity enhancement
Evaluation of the utility of the methods/protocols	Efficient measures of highly relevant site conditions	Provide scale-up to ecosystem-wide estimates	Provide direct assessment of factors affecting salmonid growth and survival
Size of project needed to provide measureable ecological response	Project size for design and prioritization	Size at which project shows signal in the ecosystem	Size of project needed to attract juvenile salmon and provide reasonable opportunity and capacity enhancement
Level and type of restoration affects rates and patterns of vegetation development, and site conditions	Project type and level of action for design and prioritization; and naturally sustainable and resilient	Project type and level of action at which project shows a signal in the ecosystem	Project type and level of action needed to attract juvenile salmon and provide reasonable opportunity and capacity enhancement
Location of site in the landscape affects the system functions	Project site selection to maximize site functions and resilience	Suite of project sites that act together to produce a signal in the ecosystem; distance between the site and the estuary affects function to the estuary	Suite of project sites which act together to produce a signal in salmonid populations
Nonlinear change in floodplain area with increase in levee breaches	Fewer breaches needed to restore near-maximum floodplain-wetted area	Exchange of species and materials between sites maximized with less than full breaching of all potential sites	Access and benefits of habitats in the ecosystem to salmonids maximized with less than full restoration of historical floodplain area
Key factors that need to be developed to maximize the rate of development and production of benefits	Hydrology, elevation, and size drive vegetation development and initiation of processes	Location of sites in the ecosystem affects relative impact on the ecosystem	Water level, driven by tidal hydrology and river flow, determine the active wetland/floodplain area that supports salmonids
Length of time the restored habitat will provide desirable benefits	With restoration of natural habitat-forming processes, should last at least 50 years; development to full functioning may take a decade to centuries	Duration of functioning within the ecosystem is tied to both the individual sites and the synergies among sites	Benefit to salmon is linked to site and ecosystem functional life; duration of restoration projects should provide benefits long enough to affect salmon populations

Table 6.3. (contd)

Finding	Inferences for the Site	Inferences for the Ecosystem	Inferences for Salmonids
Implications for restoration of the riverscape from Bonneville Dam to the mouth	Sites provide functions near (proximal) the site and to the broader ecosystem (distal)	Suites of sites provide functions to the broader ecosystem (extensive)	Restoration of functions throughout the riverscape should have at least some salmon through direct contact, processing of water properties, and by export of materials

6.3.3 Landscape Change

Watersheds that connect directly with the Columbia estuary are important to consider in an analysis of estuarine condition. Besides the obvious contribution of freshwater, watersheds provide organic matter, sediments, nutrients, spawning habitat for fish, etc. Thus the watershed makes up a critical component of the landscape of the estuary by contributing the flow of energy, species and other materials. A degraded watershed can introduce abnormal amounts of sediments, nutrients and contaminants that can affect the quality of an otherwise undisturbed estuary. The National Research Council (1992) found that the ability to successfully restore a site is significantly dependent on the degree of disturbance of the landscape within which the site is located. Therefore, improvement through restoration of the habitats in the estuary must be viewed in the context of the watersheds that contribute to them.

The habitat change analysis presented in Section 6.2.1 showed clearly the large-scale alterations in habitat distribution and land cover since historical records. Ke et al. (2012) examined changes in forest cover over a shorter and more recent time period using a geospatial change analysis based on NOAA Coastal Change Analysis Program (C-CAP). They found that forest cover in tributary watersheds to the estuary declined by 190.2 km^2 between 2001 and 2006 because of land conversion and infrastructure development (Ke et al. 2012). They reported that forest cover declined in the contributing watersheds of all reaches, with the exception of reach E (rkm 119−137), which saw a 10-km^2 increase (Figure 6.5). Watersheds contributing to reaches A (rkm 4−23) and B (rkm 23−61) showed more intensive forest loss than other reaches: forest coverage decreased from 66.1% (409.2 km^2) to 61.4% (379.9 km^2) in reach A watersheds and decreased from 56.4% (850.2 km^2) to 51.5% (776.1 km^2) in reach B watersheds. In contrast, very small declines in floodplain forest cover documented between 2001 and 2006 were detected in most reaches (<0.8 km^2), while a somewhat greater decline was seen in reach C (rkm 61−103; 3.1 km^2).

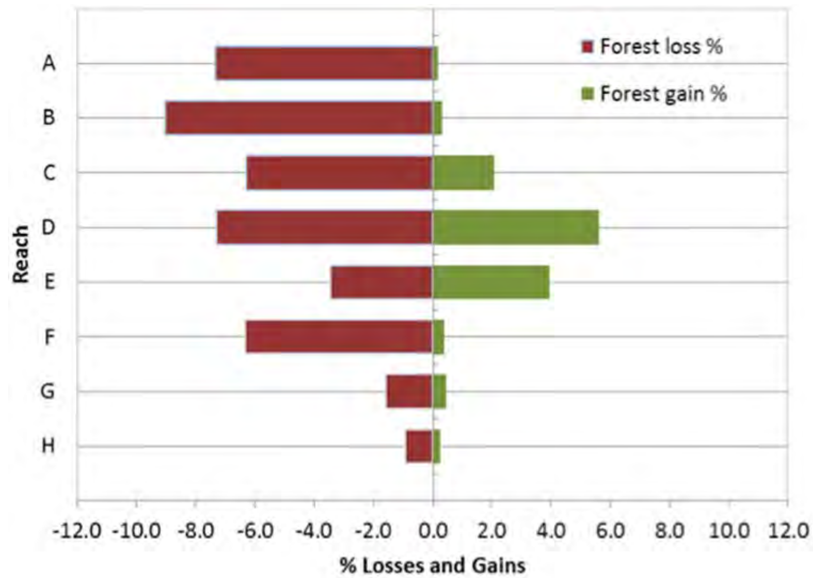


Figure 6.5. Percentage of Forest Gains and Losses in Watersheds Contributing to the Eight Estuarine Reaches (Figure A.3 from Ke et al. 2012)

These results further quantify, over a shorter time scale, the results presented in Section 6.2.1, and emphasize the fact that watersheds continue to be rapidly altered. The evidence from the habitat monitoring and reference site studies at sites located throughout the system indicates that the habitat position, size, vegetation cover and composition, channel morphology, elevation, and accretion rates vary spatially and interannually. The watersheds affect these elements, as do man-made changes to the estuarine system such as navigation improvement and diking and filling of wetlands (Sherwood et al. 1990). Restoring the estuary will become increasingly difficult if the watersheds that are tributary to the estuary main-stem continue to be degraded. This is because the habitat forming and maintaining processes necessary to the development and maintenance of restored sites will be further compromised.

6.3.4 Comparison to the CREDDP Studies

The CREDDP studies represent a comprehensive “picture” of conditions on a suite of ecosystem elements as of the period from 1978 through 1984. As a goal, CREDDP was to increase the understanding of the hydrology, sedimentology, and ecology of the estuary in order to improve the information base for managing natural resources and planning development (Simenstad et al. 1990a). The study area ranged from the river mouth (rkm 0) to approximately rkm 75. The elements studied were as follows:

- sedimentary geology
- circulation, density distribution and neap-spring transitions
- salinity and circulation modeling
- energetics and sedimentary processes
- primary production
- plant and detrital biomass
- particle transport
- community structure, distribution, and standing stocks of benthos, epibenthos, and plankton

- species composition, distribution, and invertebrate prey of fish assemblages
- consumption processes and food web structure.

The analysis of the information was synthetic, in that the authors focused on processes as well as structure. Among other findings, the historical change analysis by Sherwood et al. (1990) found that all of the elements listed above had been significantly altered primarily by flow modifications, trapping of sediment above the dams, diking and filling of floodplain wetlands, harvest of floodplain forests, urban and rural development in the floodplain and surrounding uplands, and fishing pressure. The shallow areas in the lower, predominantly tidal, portion of the river have seen net sediment accretion, whereas upstream areas are generally sediment starved. The jetties and dredged material islands have altered sedimentation processes in the former delta and along the coast north of the river mouth. Loss of habitat had resulted in an estimated 82% reduction in total shallow-water emergent plant production and a 15% loss in benthic microalgal production on tide flats. Among the most striking findings was significant alteration of the estuarine food web through the reduction in marsh macrodetritus input, and an increase in planktonic organisms primarily produced in the reservoirs upstream of the dams.

Although not studied by the CREDDP, contaminants have generally increased through time. Based on sediment cores in the Youngs Bay region of the estuary, sediment grain size distributions changed in about 1940, along with a shift from primarily benthic freshwater diatoms early in the century to planktonic species after 1940 (Peterson et al. 2003). Similar but less dramatic shifts were seen in cores from Grays Bay and Claskanie Flats. Concentrations in lead and mercury showed increases after 1920 and 1960, respectively. Contamination in sediments and the water column remain a significant concern (Morace 2012). Legacy contaminants such as (DDT) and PCBs persist in the sediments. Contaminants such as PBDEs are emerging (LCREP 2010). PAHs remain a persistent contaminant. Wastewater-treatment plant effluent and stormwater runoff contribute complex mixtures throughout the lower river (Morace 2012).

6.4 Climate Variation Change and Restoration

Climate change threatens the quality and function of the LCRE by altering three aspects of the system: river flow, water temperature, and sea level. The National Research Council Columbia River basin report concluded that "…flows and the temperature requirements for salmonid resources and the threatened and endangered stocks should be evaluated in the context of historic and potential future variability and change in both water temperature and stream flow" (NRC 2004, page 152). However, the NRC could not resolve the actual dynamics (i.e., periodicity, volumes, water levels) of the flow regime associated with climate change scenarios because of uncertainties associated with the models. In the estuary, flows affect water level and thus access by juvenile salmon to productive shallow-water floodplain habitats for feeding and rearing. Diking and flow regulation have resulted in a 62% reduction of the shallow-water habitat area accessible to juvenile salmon (Kukulka and Jay 2003). With lower flows, opportunities to access to habitats would be further limited.

Temperatures in the Columbia River basin have increased steadily over the past several decades. Scenarios of future changes show the temperature and flow conditions moving to those observed during warm phases of the Pacific Decadal Oscillation during the last century. This means that fish will be experiencing warmer water temperatures during all months. Further, warmer water temperatures may constrain the time period suitable for juvenile salmon in shallow-water habitats. Juvenile salmon appear

to vacate these habitats when water rises about approximately 19°C (Bottom et al. 2011; Roegner et al. 2010). This temperature threshold is generally reached in mid-July, when young salmon are still abundant in the system. Because of predicted changes in flows and temperatures, Barnett et al. (2004) concluded that residents and industries will have to face the choice of having water for summer and fall hydroelectric power or spring and summer releases of salmon runs; the river cannot be managed for both.

Restoring riparian vegetation has been proposed to mitigate high temperatures. Burges (2010) noted that the cooling potential of riparian vegetation is likely to postpone stressful temperatures for salmonids in the Wenatchee river main-stem. Roegner et al. (2010) found that water temperatures in the Grays River remained cooler than those in the estuary proper and Kandoll Farm restoration site located in the tidal portion of Grays River. They speculated that the warm water in the main-stem had a greater influence on water temperatures at Kandoll Farm than at Grays River proper. Pre-restoration water temperatures at this site were frequently several degrees warmer compared to the reference site channel. After restoration, the difference between the two locations was generally less than 1°C. A large proportion of water entering Grays River in summer flows upstream, being forced by tides. Air temperatures in the tidal slough at the Kandoll Farm reference site, which is dominated by a dense canopy of overhanging trees and shrubs, were consistently lower than those in the Kandoll Farm restored site (H. Diefenderfer, unpublished data). These results suggest that the effect of climate warming on these systems can be postponed by restoring direct connections between the cooler river and the restoration site, and that restoring very dense overhanging riparian vegetation can reduce temperatures in tidal channels. However, as warming continues, the effectiveness of hydrologic reconnection and dense shade to mitigate heating of the water is uncertain. The limited results available so far suggest that establishing hydrological connection is fundamental to restoring the resiliency of these shallow water habitats to warming, and thus maintain cooler waters conducive to salmonid use (see Section 5.10). We recommend that this concept be evaluated more fully in the Columbia estuary.

Besides temperature-driven shifts in populations, the rates and dynamics of many processes associated with shallow-water habitats are affected by temperature (e.g., Doney et al. 2012). The vegetation assemblage productivity, prey resource production, nutrient cycling, organic matter decay, benthic respiration (i.e., oxygen depletion), etc., are strongly influenced by temperature, and will be altered with warming. Based on the studies of water properties exchanges between the Kandoll Farm restoration site and the river, Woodruff et al. (2012) showed that the Kandoll Farm restored wetland was actively processing inorganic nutrients and organic matter. It appeared to be a sink for total organic carbon, silicate, and total suspended solids, and a source of nitrate during spring. Annually, the system was a source of organic carbon to the floodplain and broader estuary. Insects produced in the site, many species of which are important prey items for juvenile salmon, were exported along with the organic matter. How temperature affects processes in the shallow-water areas of the estuary is unstudied. Finally, based on an analysis of a variety of physical (e.g., stratification, mixing) and biological factors (e.g., chlorophyll), Roegner et al. (2011b) concluded that the interaction between stream flow and strong coastal upwelling in the vicinity of the mouth of the river have strong ecological ramifications for riverine, estuarine, and oceanic ecosystems in the Pacific Northwest. Climate change-driven changes in these factors result in ecological changes throughout these ecosystems.

Sea-level rise presents the third climate change-related threat to the estuary and salmon. Jay (2009) predicted that sea-level rise would result in increased tidal amplitudes in the northeast Pacific. Although records show that the mean sea level has been steady at Astoria since the 1920s, this is explained by the fact that tectonically driven rise in land elevation is keeping pace with eustatic sea-level rise at Astoria.

Further upstream, the land surface is falling relative to sea level. As the rate of sea rise accelerates, the Corps (Portland District COE 2012, unpublished) predicts, under the intermediate rise rate scenario, an approximate 1-ft (0.30-m) rise in sea level by 2060, and an approximate 3-ft (0.91-m) rise by 2100. In general, a rising sea has been shown to destroy intertidal wetlands that cannot keep pace through natural accretion processes. In areas of coastline such as the Mississippi River delta, where wetland subsidence and lack of sediment supply exacerbate the effect of eustatic rise, large areas of wetlands are lost annually (e.g., Day et al. 2007). The same would be expected in the Columbia estuary. Borde et al. (2012) showed that floodplain wetlands are restricted to an approximate 1-m elevation range. Hence a rise of 0.3 to 0.9 m would affect much of the wetlands existing in the lower tidal-dominated portion of the estuary. In the lower estuary, sea-level rise would be accompanied by salinity intrusion into freshwater wetlands, further stressing systems and ultimately causing floral and faunal shifts. Wetlands are expected to retreat upland as the sea rises, but impediments such as levees, roads, steep topography, and developed infrastructure will prevent the retreat in many places in the system. Restored floodplain wetlands have been shown to accrete sediment and organic matter faster than existing wetlands in the estuary (Thom et al. 2012). However, the intermediate rise rate scenario would exceed accretion rates in all wetlands sampled so far.

6.5 Conclusions

The physical changes, including floodplain development, dredging of the navigation channel and harbors, and flow regulation, significantly altered the historical geomorphic and ecological state of the LCRE prior to the CREDDP studies (Table 6.1). However, the rate of physical alteration has apparently slowed compared to the late 19th and early 20th century. Physical changes are still occurring. The navigation channel was deepened (1−3 ft) early in the present century, and channel maintenance, including dredge material disposal in the estuary is conducted periodically. The habitat complexes within the present floodplain form a highly altered mosaic compared to historical conditions (Simenstad et al. 2011). Non-native species are abundant and dominate vegetation, plankton, fish, and benthos assemblages. Very few "historic" (i.e., late 1800s) wetland habitats remain in the system (Borde et al. 2012). The biological communities and geomorphology of the system are structured by natural disturbances (e.g., floods), with evidence that the habitat mosaic shifts spatially when forced by hydrological conditions and other controlling factors (Simenstad et al. 2011; Borde et al. 2012). Pile dikes, designed to maintain the navigation channel location and depth, have resulted in deposition of sediments and the formation of shallow-water habitats (Kassebaum and Moritz 2012). The rate of introductions of non-native species may be decreasing, but this is difficult to discern. Data show an expansion of invasive, highly competitive, non-native species such as reed canarygrass. There is a legacy of contamination in sediments. Contamination of water and sediment from persistent chemicals is increasing and is of significant concern. Through alteration in river flow dynamics and volumes, increases in water temperature, and sea-level rise, climate change is expected to affect the ecological processes of shallow-water habitats, and the capacity of the habitats to support young salmon.

Restoration projects focused on floodplain habitats have increased over the past decade (LCREP 2010; Sagar et al. 2012). These actions are showing immediate benefit to juvenile salmon by providing access to habitats as well as processes supportive of ecosystem services (Table 6.3) of benefit to the entire estuary. Further, natural breaching of levees and dikes has opened areas of former floodplain habitats (Diefenderfer et al. 2010). The land surface formerly behind the levees had obviously subsided and most sites remain dissimilar to nearby reference sites even after several decades (Borde et al. 2012). Hence, the full return of floodplain habitats to their historical state will be protracted, especially those dominated by

tidal forested swamps. Yet, these systems will predictably continue to provide services during development phase. Emergent marsh habitats show large changes during the first four to seven years with full development to reference conditions predicted to be on the order of 75 years (e.g., Thom et al. 2003). As evidenced in historical natural breaches, estuarine riparian and tidal forested habitats can develop within several decades of reconnection, and do have intermediate stages that are contributing services to the system (Diefenderfer et al. 2010). Net ecosystem improvement is hampered by development activities such as road construction and resource extraction in tributary watersheds draining into the lower floodplain habitats and broader LCRE.

7.0 Summary of Findings

Findings relative to each research theme are summarized below.

7.1 What are the Contemporary Patterns of Juvenile Salmon Habitat Use in the Estuary and Factors that Potentially Limit Salmon Performance?

- Six species of salmon and anadromous trout were identified in the shallow-water habitats of the estuary: Chinook salmon, coho salmon, chum salmon, sockeye salmon, steelhead, and coastal cutthroat trout. However, the primary species inhabiting the shallow-water environments investigated are Chinook, chum, and coho salmon.
- These species and their various stocks display variations in juvenile life history characteristics. Chum salmon are primarily fry migrants that enter the ocean at ≤60-mm FL. Chinook and coho salmon are present in both subyearling and yearling life history stages. All life history types of Chinook salmon are found year-round throughout the habitats sampled and enter the sea at all sizes. Coho subyearlings are found primarily in tributary reaches and migrate to the sea as yearlings.
- Species and stocks (especially of Chinook salmon) have distinct migration periods. In shallow water areas, subyearling Chinook salmon can be found year-round, but their density peaks in April–June and drops to low levels after July; yearling Chinook salmon migrate from March through May. Chum salmon move through the system from February through May. Most yearling coho are found in the system in May.
- For Chinook salmon, the proportion of fish with hatchery marks has increased in recent years, and now clearly indicates the overall predominance of hatchery-reared fish at most main-stem sampling sites. However, the many fry-sized fish found in shallow-water habitats are unmarked and are likely wild fish. Shallow-water areas are particularly important to this life history type.
- Restrictions to habitat opportunity may be limiting salmon recovery. Hydrological barriers exist on numerous waterways, reducing or excluding salmon entry. Oxygen concentrations become low in systems with restricted exchange during summer and near the mouth of the estuary during strong upwelling winds. Temperatures increase past the criterion thought to induce stress (19°C) system-wide during June through September every year. Both low DO and high temperature may force salmon from shallow-water systems and induce stress that can affect growth and survival. Restoration that reconnects hydrological links has been shown to improve these elements of physical habitat opportunity.
- The habitat capacity of studied wetland systems appears to be relatively positive. These systems produce large amounts of insect and amphipod prey, highly favored and energy-dense salmon food, which are used in situ and also after export to the surrounding environment. Based on diet overlap and prey productivity, competition between salmonid species and between salmon and other fish species appears to be relatively low. Fish predation on salmon within systems has not been specifically studied, but potential predators exist in some studied sites. However, bird predation on salmon within the LCRE is very high. At present, the degree of avian predation on salmon in wetlands has not been determined.

- Studies of salmon performance have focused on feeding, growth, and habitat residency. Juvenile salmon feed heavily on insects and amphipods in restoring and reference wetlands, and their gut contents are generally high. Growth rates derived from various methods range from 0.3 to 1.62 mm/d. These data indicate salmon are benefitting from wetland habitats. However, overall losses of wetland habitats may be reducing salmon performance on the estuary scale.
- The residence time of salmon in wetland habitats varies by life history stage, location, and season, but it is clear that some salmon remain in or near wetland channel locations for weeks to months.
- Concentrations of organic contaminants, PCBs, PAHs, and PBDEs may present significant health risks for juvenile salmon.

7.2 Do Factors in the Estuary Limit Recovery of At-Risk Salmon Populations and ESUs?

- Salmon habitat surveys in the Columbia River estuary since 2002 have provided new data on the stock-group affiliations of juvenile Chinook salmon. The results from new tagging techniques, otolith chemical analyses, and an improved genetic baseline provide a first glimpse of stock-specific habitat associations, salmon life histories, and performance of juvenile Chinook salmon within the estuary. Much less information is available regarding the stock or population origins of other salmonid species in the estuary.
- Different sampling methods (gear, locations, time periods) select for different salmon stocks and life history types such that no single study can provide a complete picture of salmon behavior or stock composition within the estuary. Most RME studies are designed to sample shallow-water and near-shore areas, targeting the habitat types that have been most intensively modified by historical development and that are a primary focus of estuary restoration.
- Methods for sampling deep channels (e.g., purse seine, pair trawl, acoustic–tag monitoring) tend to select higher proportions of large yearlings and hatchery fish than the beach-seine samples collected in shallow near-shore habitats. Stock composition also varies somewhat between deep and shallow habitat methods, including a higher prevalence of interior spring and fall run stocks in lower estuary purse-seine collections than in nearby beach-seine samples.
- Columbia River Chinook salmon stocks are not uniformly distributed in space or time, but they exhibit characteristic patterns of migration and habitat use. All genetic stock groups of Chinook salmon except interior spring stocks frequent shallow habitats of the lower and mid-estuary, but lower Columbia River fall Chinook are most abundant. Greater proportions of upper Columbia River summer/fall, Willamette River spring, and interior (i.e., Snake River, Deschutes River) fall Chinook stocks are represented in upper estuary reaches. Unpublished data from a recent series of estuary-wide genetic surveys reinforce these patterns, and suggest that variations in overall stock composition are consistent between years and at an estuary-reach scale.
- At a site scale, in contrast, genetic survey results are quite variable, and no consistent differences in stock proportions are apparent among different shallow-water habitat types within the same estuary regions.

- Most Chinook stocks are able to express estuary-resident life histories and benefit from estuarine prey and growth opportunities. However, some at-risk stocks are poorly represented in field collections, and additional sampling will be needed to improve understanding of their estuarine habitat needs and performance.
- Stock-specific habitat use and limiting factors have not been documented as fully in the upper estuary as they have been in the lower estuary. Additional monitoring is needed to validate the general patterns that have been described at a reach scale and to determine whether habitat associations in the upper estuary are stock-specific.
- On average, estuary residence time is inversely related to fish size, but considerable variation exists among the stock groups and life history types targeted by different sampling methods, survey periods, and locations. As estuary sampling effort has increased, so has understanding of life history variations within and among stocks:
  - West Cascade Tributary fall and Spring Creek group fall stocks in the lower estuary are represented primarily by fry and fingerling migrants. However, less abundant stock groups, including West Cascade tributary spring and Willamette River spring stocks, express a wider range of juvenile sizes and ages at migration.
  - Most large hatchery fish are believed to migrate from Bonneville Dam to the river mouth in a few days. However, PIT monitoring indicates that some hatchery fish remain in the estuary for weeks and use lower-estuary wetland channels before migrating seaward.
  - Upper Columbia River stocks that enter the estuary at relatively large sizes might be expected to migrate rapidly through the estuary. However, otolith-derived residency estimates for the brackish portion of the estuary averaged 82 days for one sample of upper Columbia River summer/fall salmon (N = 9) that averaged 88 mm FL at estuary entry.
  - Spring Chinook salmon are typically considered yearling migrants that enter the estuary in spring and move rapidly to the river mouth. However, West Cascade tributary and Willamette spring Chinook stocks produce both yearling and subyearling migrants.
  - Fall Chinook stocks are typically considered subyearling migrants. However, acoustic tagging results indicate that representatives of both spring and fall Chinook salmon stocks reside for weeks in shallow areas near the Sandy River delta in winter and spring. Some of these individuals may over-winter in the upper estuary.
- Despite examples of unexpected variation in some stock groups, life history diversity in Chinook salmon appears simplified relative to historical patterns, when pulses of subyearling migrants entered the estuary late into the summer and fall. Multiple factors could account for apparent reductions in life history diversity, including upriver population losses, intensive hatchery production of a few selected phenotypes, and loss of shallow rearing opportunities within the estuary.
- RME activities within the estuary have not quantified the estuary's contributions to adult returns among different ESUs. The effects of improved estuary growth or survival for subsequent life stages of Columbia River salmon remain poorly understood. The results of estuarine studies must be placed in a broader life-cycle context to estimate the estuary's contributions to population viability.
- No one research tool or design will be adequate to interpret the estuarine life histories or quantify the estuary's contributions to all Columbia River stocks. A combination of approaches specific to the sampling challenges and life history pathways of each ESU will be required.

7.3 Are Estuary Restoration Actions Improving the Performance of Juvenile Salmon in the Estuary?

- Restoration in the LCRE can offer positive benefits to juvenile salmon in terms of opportunity, capacity, and realized function.
  - Hydrologic reconnections can increase opportunity for fish to access restored sites, as noted at Crims, Kandoll Farm, and Ft. Columbia.
  - In terms of evaluating capacity, improvements in water temperature were noted at Kandoll Farm and South Slough while improvements in the abundance of certain prey items were noted at Crims Island.
  - A positive benefit of realized function was observed at Crims Island by examining residence time and foraging success of juvenile Chinook salmon.
  - An additional assessment of inferences made with respect to ecosystems processes and restoration in the LCRE is found in section 6.3.2.
- Of the 42 aquatic restoration sites that have been completed in the LCRE since 2001, only a small fraction (n=9) included AE monitoring that directly addressed elements relevant to juvenile salmon ecology (i.e., opportunity, capacity, and realized function). Three of the nine restoration projects lacked reference sites and before and after datasets.
- Seven of the nine sites reviewed occurred in the lower 90 rkm of the estuary and most were concentrated in reaches A and B (rkm 4−61). Reaches C−H are underrepresented in terms of AE research in the LCRE. There were no AE sites located in reaches E and G.
- Some sites (e.g., Tenasillahe Island and Vera Slough) provided examples where, despite actions aimed at improving hydraulic connectivity via tide-gate replacement and/or retrofits, the functionality of these tide gates may continue to impede access to sites, at least in comparison with reference sites.
- Most studies examined water temperature and found that thermal conditions exceeded EPAs recommended water temperature of 19°C for juvenile salmon rearing preference (EPA 2003) during summer months. In addition, some AE sites exceeded 19°C during spring months. Some studies reported sub-optimal thermal conditions in reference sites as well. Qualitative comparisons suggest the warm water temperatures observed in restored sites during summer months are similar to those observed in shallow water habitats across the LCRE.

7.4 What is the Status of the Estuary? Are Estuarine Conditions Improving or Declining?

- Physical changes, including filling of the floodplain, dredging of the navigation channel and harbors, and regulating flow significantly altered the historical geomorphic and ecological state of the LCRE system prior to the CREDDP studies. The rate of physical alteration has apparently slowed compared to the late 19th and early 20th centuries.
- Habitat complexes within the present floodplain form a highly altered mosaic compared to historical condition, and very few historic (i.e., late 19th century) wetland habitats remain in the system.
- Based on an analysis of levels of stress associated with diking, overwater structures, land conversion, etc., at both site and watershed scales, the LCRE ecosystem is “moderately stressed” compared to

conditions prior to dam construction, forest harvest, diking, etc. Most altered reaches include Portland, Vancouver, and Longview.

- Data show an expansion of invasive, highly competitive, non-native species such as reed canarygrass.
- Through alteration in river flow dynamics and volumes, increases in water temperature, and sea-level rise, climate change is expected to affect the ecological processes of shallow-water habitats and capacity of the habitats to support juvenile salmon.
- Understanding the link between hydrodynamics and vegetation is critical to designing and predicting the outcomes of restoration projects, understanding interannual variation in assemblages, and predicting the effects of larger scale changes (e.g., climate change).
- The number of restoration projects focused on floodplain habitats has increased and positive effects have been seen on site-scale and ecosystem-scale habitat conditions. These effects benefit juvenile salmon and the entire estuary through export and exchange of organic matter and prey export.
- Literature indicates that biodiversity can be a strong regulator of ecosystem function.
- Species area curves revealed higher species richness in restoring sites in the estuary as compared with pre-restoration species richness. Natural breaching of levees and dikes has occurred over the past several decades and these systems contain wetland assemblages and harbor fish including juvenile salmon. Although the full return of floodplain habitats to their historical state will be protracted, these systems will predictably continue to provide services during this development phase.
- Processes required to form and maintain floodplain habitats are generally restored once natural hydrodynamics are re-established at a site.
- Processes including water temperature modulation, sediment accretion, vegetation structure development, fish access, and flux of organic matter were improved rapidly (over the first five years) following hydrological reconnections.
- Net ecosystem improvement through restoration of floodplain habitats is potentially hampered by recent human impacts (e.g., road construction and resource extraction in tributary watersheds serving the lower flood plain habitats and broader LCRE).
- Climate change threatens the quality and function of the LCRE by altering river flow, water temperature, and sea level. Restoring wetlands should mitigate effects on water temperature through enhanced water exchange and shading of channels by dense vegetation.

7.5 Summary of Findings - Conclusion

In this section we summarize the findings from above that we believe are most relevant to the CEERP objectives and decision making for restoration and Research Monitoring and Evaluation.

7.5.1 Research, Monitoring, and Evaluation

Relative to the information available before 1990, we concluded that considerable progress has been made understanding the habitat needs of juvenile salmon and the ecology of the Columbia River estuary. Most impressive are the results of an intensified research and monitoring program that has amassed a wealth of new data within the last decade, including information about the following:

- the estuarine habitat associations and life histories of juvenile Chinook salmon, and the use of wetlands and other shallow, near-shore habitats for rearing
- the growth rates, residence times, and food webs of subyearling Chinook salmon and various stressors (e.g., temperature, DO, disease) that influence salmon within the estuary
- the genetic affiliations of individual Chinook salmon within the estuary and the distinct temporal and spatial patterns of estuary use by juveniles from different ESUs
- life history variations expressed by juvenile salmon that contradict the "stream-type" and "ocean-type" dichotomy traditionally ascribed to spring and fall Chinook salmon, respectively (e.g., the presence of subyearling spring Chinook migrants from lower Columbia River and Willamette River ESUs and overwintering subyearling migrants from lower and upper Columbia River fall Chinook ESUs).

Among the key findings of recent research, monitoring, and evaluation in the Columbia River estuary are the following:

- Although all salmonid species resident in the Columbia River watershed were detected in shallow water environments, Chinook, chum, and coho salmon were by far the most numerous. Chinook and chum were mainly fry and subyearling migrants, with smaller numbers of yearling Chinook salmon also present. Coho yearlings predominated at main-stem sites and subyearlings were found at some tributary and backwater habitats. Species and stocks have distinct migration periods that determine when and where restoration actions will affect them. While the majority of Chinook and coho were hatchery reared, most chum and many fry-sized Chinook salmon found in shallow-water habitats are likely wild fish requiring protected rearing habitats. Most Columbia River salmon stocks are capable of extended periods (i.e., weeks or months) of estuary rearing, but the estuarine life histories and habitat associations of many low-abundance stocks, including at-risk salmon from interior basins, are not well known. Little has been done with regard to investigating the health and fitness of juvenile salmon using physiological metrics in the LCRE.
- Recent improvements in the genetic baseline for Chinook salmon have allowed monitoring programs to identify the stock affiliations of individual fish sampled in the estuary and to compare the habitat associations, life histories, and performance of salmon among different stock groups. Chinook genetics data collected throughout the estuary provide evidence that salmon stocks exhibit distinct and characteristic patterns of estuary rearing and migration. The results suggest that the habitat needs of each stock will vary according to their particular temporal and spatial pathways through the estuary. However, genetic data alone often are not sufficient to identify the geographic origins of juvenile salmon: the existing baseline is too coarse to distinguish fine-scale genetic differences (i.e., individual populations), and past stock transfers have redistributed many genetic stock groups far from their natal basins. In addition, the influence of estuary rearing habitat on adult returns is poorly understood. New study approaches have begun to address this information need, including the use of (1) otolith chemical methods to estimate the contribution of diverse juvenile life histories to adult survivors, and (2) life-cycle models to explore the sensitivities of salmon populations to estuary survival improvements. Little genetic data has been applied to other salmon species.
- Of the 42 restoration projects reviewed, only nine included AE monitoring. Of these nine, six included reference sites as well as pre-restoration monitoring, and one completed a formal before-after analysis to evaluate biotic and abiotic response after restoration. Lack of pre-restoration data

and appropriate reference sites within an AE study limits the ability to infer response resulting from restoration actions. AE research elements aimed at evaluating salmon performance (e.g., growth, residence time, foraging success) in restored sites were rare in the majority of projects reviewed. Studies of shallow-water habitats have focused on the lower reaches of the estuary (Reach A through C), where lower river stocks tend to dominate sample collections. Although recent surveys have found a greater prevalence of interior stocks in the upper estuary, the habitat requirements of stocks in this region are not well documented. In addition, AE research was largely focused on spring and summer migration periods which limit inferences that can be made with respect to the life history diversity of juvenile salmon in restored sites during other times of the year. There are a paucity of reference sites that represent historical floodplain habitats, thus limiting both the planning of restoration project designs and assessing the restoration target of restoration projects. However, naturally breached sites and fragmented historical wetlands do exist.

- For all of these metrics, the effect of climate change on the long-term sustainability of restored habitats is uncertain. Detailed synoptic studies are needed to establish a baseline of existing conditions to compare with CREDDP and help identify trends for future change.

7.5.2 Habitat Restoration

- Habitat opportunity is limited by extensive hydrological barriers, and in areas with limited water exchange, by low dissolved oxygen and/or high temperatures. Unsuitable water-quality parameters are more common during summer low flow periods. Restoration that reconnects hydrological links has been shown to improve these elements of physical habitat opportunity. However, not all hydraulic reconnections are created equal. Of the AE projects reviewed, some hydraulic reconnection projects failed to create opportunities for juvenile salmon to access sites, whereas other reconnection projects succeeded at increasing habitat connectivity and opportunity for fish to access sites. Monitoring programs to date are limited but most have documented habitat use by juvenile salmon or demonstrated the benefits (e.g., foraging success, life-history variation, and growth) of estuary rearing at juvenile life stages in shallow water habitats of the LCRE.

- The habitat capacity of "natural" and restored wetlands is enhanced by high production of energy-rich insect and amphipod prey. Insects, in particular, are produced in wetlands and shallow water habitats and contribute disproportionally to salmon diets both within wetlands and also by larger fish after export to main-stem habitats. Data to date do not indicate high levels of competition or predation within wetlands and shallow water habitats, although bird predation is a serious source of mortality for some stocks of salmonids in the saline estuary and lower river. However, salmon condition and contaminant studies suggest high variability in salmon health metrics, with unknown consequences for population resiliency.

- Monitoring to date indicates that restoring former floodplain and intertidal wetland systems to historical levels of hydrological reconnections results in rapid initial recovery of plant assemblages and ecological processes relevant to salmonids. However, quantitative relationships between structural metrics and the functional responses of salmonids have yet to be established. Very limited studies on export of marsh macrodetritus from restored sites indicate that a large proportion of the macrodetritus is exported over considerable distances to the estuary proper. Restricted hydrological reconnections are generally less effective. Habitats that develop on dredged material disposal islands contain a community that differs from natural reference wetlands, but still appear to be functional and may benefit salmon.

- Invasive non-native species, particularly reed canary grass, threaten the full recovery of the historical community structure of former floodplain wetlands being restored. Research is sparse on the ecological role of non-native vegetation species, especially the link between supporting salmon food webs (e.g., capacity) and the contribution to the broader ecosystem of marsh-derived macrodetritus.

8.0 Recommendations

From our review of the status of science in the Columbia River estuary and the conclusions outlined above, we developed two broad categories of recommendations for the CEERP— RME and restoration. The topics included in our recommendations generally follow from our evaluation of available information related to each of the four questions posed at the outset of this review:

1. What are the contemporary patterns of juvenile salmon habitat use in the estuary, and what factors or threats potentially limit salmon performance?
2. Do factors in the estuary limit recovery of at-risk salmon populations and ESUs?
3. Are estuary restoration actions improving the performance of juvenile salmon in the estuary?
4. What is the status of the estuary? Are estuarine conditions improving, declining?

Most of our analysis considered the adequacy of information to fully answer the four questions above. Accordingly, many of our recommendations suggest changes or additions to the RME program to improve subsequent evaluations of salmon and estuary ecosystem response to the CEERP.

8.1 Research, Monitoring, and Evaluation

Quantify the individual and synergistic effects of stressors (e.g., high temperature, low dissolved oxygen, and bioaccumulative chemicals) on salmon condition and survival.

Physiological measurements of salmon fitness or condition in the estuary are limited, despite measured deleterious water-quality levels and high body burdens of toxics. These estuarine conditions may be influencing juvenile salmon performance and survival at ocean entry. Methodologies are needed to investigate effects of stressors (e.g., high temperature, low dissolved oxygen, and bioaccumulative chemicals) on subsequent survival, ideally by using non-lethal sampling techniques. In addition, controlled laboratory experiments using known genetic stocks and life history stages are needed to understand intrinsic variation among salmon types. Laboratory work should explicitly examine thresholds and synergistic effects of high temperature, low DO, and toxic chemicals on salmon tolerance, behavior, and fitness for use in life-cycle and habitat modeling.

Investigate the effects of large hatchery releases on predator populations and food webs in the LCRE and their implications for at-risk salmon stocks.

Recent surveys in the LCRE have documented increases in avian predator populations and significant losses of juvenile salmon to predation. Yet, the ultimate causes or effects of these trends are poorly understood. For example, increasing predator populations could be an ecosystem-level response to the concentrated pulses of similarly-sized salmon smolts that are released from hatcheries every spring. Additional research is needed to investigate the ecological effects of hatchery programs on the estuary ecosystem, including behavioral responses of avian predators to large hatchery releases, the effects of avian predators on estuarine food webs, and the secondary effects of these ecosystem changes for at-risk salmon.

Develop a suite of physiological markers to indicate the health of juvenile salmonids, especially as they relate to benefits derived from restored habitats.

Non-lethal physiological markers indicative of salmonid performance (e.g., growth, foraging success, and condition) in restored sites may offer an alternative to current approaches, which can be costly. An ideal marker would help identify benefits juvenile salmon derive from using wetland habitats and especially restoration sites. Such markers could also be applied at sentinel sites and serve as indicators for overall health and condition of fish in the across the gradient of the LCRE.

Develop genetic or other techniques to further resolve the geographic origins of juvenile salmon found in particular regions and habitats of the estuary.

Despite considerable progress in genetic stock identification techniques over the last decade, the resolution of the existing genetic baseline for Chinook salmon is limited to approximately the ESU level. This scale is sufficient to compare general patterns of estuary use among different stock groups, but cannot be used to discern the stream origins of individuals or differences in estuary habitat use among a diversity of populations within ESUs. A higher resolution genetic baseline would benefit estuary restoration efforts on behalf of Chinook salmon. In addition, new genetic baselines would be needed to account for differences in estuary-habitat use among different stocks of chum, coho, sockeye, or steelhead. Identification of salmonid population sources could benefit from further application of otolith chemical methods if subbasins or other geographic areas with distinct chemical signatures can be readily identified and validated.

Expand surveys in upper estuary reaches to compare habitat use and performance among Chinook stocks, including stock groups that are poorly represented in most lower-estuary sample collections.

Early RME activities were focused primarily in the lower estuary, where stocks from the lower Columbia River ESU are most abundant. Recent surveys have provided evidence that higher proportions of other stock groups occur in upper-estuary reaches, although the habitat associations and life histories of these stocks are not well documented. Additional surveys are needed, particularly in reaches D–H, to determine stock-specific use of a diversity of floodplain, forested slough, and other habitat types represented in the upper tidal-fluvial region of the estuary. As in other locales, surveys should compare salmon life histories and performance (i.e., growth, foraging success, survival) among all genetic stocks, including less-abundant stocks (e.g., Willamette River spring, Deschutes River fall, Snake River fall) that may reside for extended periods in the upper estuary before migrating to the river mouth.

Develop RME methods and study designs to quantify the estuary's influence on adult returns and to estimate the effectiveness of estuary restoration for salmon recovery.

Estuary restoration and RME activities assume that improved salmon performance within the estuary will benefit survival and recovery of at-risk stocks. However, validation of this assumption will require other research methods to account for estuary linkages to the rest of the salmon life cycle. In a few tributaries it may be feasible to quantify juvenile and adult population abundances and directly estimate estuary contributions to adult returns using various mark and recapture methods. Such studies are best suited to small estuary tributaries that contain the full continuum of freshwater-tidal habitats, where salmon habitat use and life histories can be readily monitored before outmigrants disperse throughout the

main estuary. Some Oregon and Washington life-cycle monitoring programs already established in various tributaries of the LCRE could be expanded to quantify estuary contributions to adult survival and life history variability.

For interpreting estuary contributions to interior populations that enter the main-stem Columbia far upstream of the estuary, adult otoliths can be analyzed to reconstruct the sizes and times of estuary entry among those juveniles that survive to return. Otolith chemical techniques for life history determinations have progressed in recent years; however, additional improvements are needed to distinguish juvenile rearing in the tidal-fresh estuary from rearing periods in nontidal freshwater areas. The understanding of estuary contributions to salmon recovery would also benefit if otolith monitoring can be expanded to compare juvenile life histories across ESUs and to assess interannual life-history variations within selected ESUs. Finally, life-cycle modeling is an important tool to evaluate the relative sensitivities of each ESU to survival improvements that could result from restoring estuary habitats. However, the resolution of most existing models is relatively coarse and must be improved to explicitly account for the estuarine phase of salmon life cycles.

Develop a quantitative understanding of relationships between structural metrics and functional responses of salmonids in restored habitats.

Structural conditions (e.g., water surface elevation, vegetative cover, channel morphology) measured as part of AE research are informative from the perspective of tracking ecosystem response; however, there is not yet a direct link between these metrics and the benefits salmonids derive from restored sites. Developing a quantitative understanding of the relationships between key structural/habitat conditions and salmonid performance (e.g., attributes of growth, foraging success, residence time) will facilitate the development of ratio estimators and numerical models which will strengthen the ability to predict salmonid performance outcomes and economize future AE research.

Evaluate long-term, stock-specific responses (e.g., density, growth, conditions, and life history diversity) of juvenile salmonids to CEERP actions at landscape and estuary-wide scales.

A direct linkage has yet to be demonstrated between restoring ecosystems in the LCRE and benefits to listed salmon and steelhead stocks within the upper Columbia, Snake, and Willamette (UCSW) basins. In addition, regional efforts do not include a high-level indicator for measuring and tracking life history diversity and juvenile salmonid density, two of the primary tenets of the Fish and Wildlife Program. Establishing sentinel zones within the LCRE, where juvenile salmon and steelhead would be routinely monitored in shallow (i.e., beach seine) and deep water habitats (i.e., purse seine), would permit systematic tracking of key salmon response variables (e.g., density, growth, fish condition, and life-history diversity) and incorporating these responses into CEERP actions to assess landscape and estuary-wide benefits to listed stocks.

Investigate whether competitive interactions between hatchery and natural origin (NO) salmon significantly influence the performance of at-risk stocks in shallow-water estuary habitats.

Hatchery programs substantially influence salmon abundances and size-dependent patterns of salmon habitat use and residency throughout the estuary. Although hatchery and NO salmon distributions overlap to varying degrees, the effects of hatchery fish on estuary habitat selection and the performance of at-risk stocks are unknown. Among important uncertainties are hatchery influence on the feeding

behavior and foraging success of NO juveniles, the behavioral response of NO fry or fingerlings to larger size classes of hatchery-reared salmon, and hatchery influence on habitat capacity and the growth potential of NO salmon in selected estuary habitats.

Further refine the degree of hydrological connection that is necessary to allow for maximum salmonid access and maximum development of natural habitats and associated ecological functions.

At present most tide gate retrofits have demonstrated relatively poor functionality and limited access. If tide gate retrofits are to be considered further, the effectiveness of these options needs to be better understood. Retrofitting sites with tide gates that support greater access to sites by salmonids and the development of wetland habitat structure and functional processes (e.g., export of organic matter) needs further research to inform design, placement, maintenance, and long-term utility.

Include marsh macrodetritus export in monitoring programs at restored sites to better quantify the cumulative effects of multiple restoration projects on the ecosystem.

CEERP goals include offsite effects of restoration sites. Further it appears that exported prey is being consumed by juvenile salmonids in the main-stem of the estuary. To date, few studies document export of organic matter produced in the wetlands to the broader ecosystem. Export of both marsh macrodetritus and associated insects may be a very important response of the CERRP restoration program, but more quantification of this response would help in refining the cumulative effect of restoration projects on the broader ecosystem in support of the salmonid food web. This would include refinements such as the size of the site versus the among produced and exported, the effect of elevation on detrital export, and the effect of flood versus non-flood events on exchange of materials and chemicals (e.g., nutrients, dissolved organic carbon)

Improve understanding of the role of non-native vegetation (i.e., reed canary grass) relative to prey production for salmonids, macrodetritus production, contributions to the broader estuarine ecosystem, and effects on habitat biodiversity.

At present, many sites, especially those in the middle reaches of the estuary, are dominated by reed canary grass. Eradication is very difficult, and may not be required if this species is shown to have limited detrimental effect on CEERP objectives. To date there are very few studies that evaluate the ecological role of this species. This research will help decide whether further actions are needed to sustain support for salmonids and other ecological processes in areas susceptible to development of dominant stands of reed canary grass.

Investigate further the use of dredged material disposal to create and maintain habitats that are strongly functional for salmonid support and ecological functions.

Although vegetation has developed on dredged material islands, the actual suitability of these habitats for fish is uncertain. Investigations as to the design of dredged material placement specifically to encourage sustainable functional habitat development is needed if creation is to be considered a viable strategy within CEERP. These investigations should include the need for continued maintenance and re-nourishment of created sites.

Evaluate the response of restored habitats to climate change including effects of sea level rise and flow alterations and water temperature.

As sea level rises and water releases change, it is unclear how restored floodplain wetlands will respond. Questions important to CERRP include:

1. Will space be available for these habitats to shift to higher elevations with rising sea level?
2. What will alterations in flow alterations do in terms of supporting wetland processes and access by salmonids?
3. Will water temperature increase, reducing the effective area of restored habitats for salmonids?

Investigate whether upland development is impacting restored floodplain wetlands or the broader estuarine ecosystem quality.

The degradation of uplands could inhibit development of restored floodplain wetlands and shallow water habitat in the estuary. Spawning habitat in many reaches has been detrimentally impacted by upland development, thus limiting overall stock recruitment. It is clear that the health of the landscapes surrounding the restoration projects will affect the ability of restored wetlands to develop naturally and quickly and for functions of these wetlands to be maintained for salmonids over the long-term.

8.2 Restoration

Our restoration recommendations concern general needs for restoration planning and assessment as suggested by our review of salmon habitat use and performance in the estuary. We have not addressed the finer details of restoration-project design, which are beyond the scope of this review.

Evaluate salmonid performance in restored sites across multiple spatial and temporal scales.

Restoration planning must carefully weigh project goals against expected outcomes. Increasing access at a site does not necessarily infer a benefit to salmonids if habitat capacity is poor, nor does improving capacity if there is no access. Benefits to juvenile salmonids from restoration action are best ascribed by examining performance metrics that include attributes of growth, residence time, and foraging success. Furthermore, AE research has largely focused on spring-summer migration periods which do not provide a complete understanding of how juvenile salmon respond to restored sites during other time periods. Inferences regarding benefits of restoration actions must be made within the context of pre-restoration conditions, comparisons to carefully selected reference and/or control sites, and at site and landscape scales over short (1-3 yr) and long (5-10 yr) time frames.

Develop strategies for estuary habitat restoration that explicitly account for the temporal and spatial pathways of different salmon stocks and life-history types.

Recent genetics survey results indicate that different stock groups exhibit characteristic seasonal and spatial patterns of estuary habitat use, reflecting their geographic origins and their hatchery or natural rearing histories prior to estuary entry. Such results imply that not all stocks will benefit similarly from a particular restoration site or project design. Management agencies must plan restoration strategically to account for the broader (i.e., landscape) distribution of habitats necessary to support the varied migratory and rearing pathways of diverse salmon stocks. Restoration proponents should define the particular

stocks and life history types that are intended to benefit from a restoration action, and effectiveness should be evaluated relative to these objectives. New genetic and life history data will continue to improve understanding of stock-specific habitat needs throughout the estuary. Nonetheless, a requirement to specify salmon-stock objectives will promote learning and adaptation by requiring that available genetics data are considered during project planning and by setting measurable goals for evaluating future restoration success.

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Appendix

Detailed Recommendations for Ecosystem Restoration in the Lower Columbia River and Estuary

Appendix

Detailed Recommendations for Ecosystem Restoration in the Lower Columbia River and Estuary

Table A.1 and Table A.2 were the result of an effort by us to capture the full range of potential recommendations that emerged from the development of this report. We collectively developed the justifications for each recommendation and indicated the major relevant report sections that formed the source of the recommendation. Finally, we developed narratives to identify the specific relevance of the recommendations to the CEERP objectives, along with the relevant reasonable and prudent actions (RPA). Table A.1 and Table A.2 essentially capture our thinking at a point when the report findings were fresh in our minds. From this effort, we culled the specific recommendations presented in the main body of the report (Section 8.0). We also provided more focused justification in that section.

Table A.1. Recommendations for Research, Monitoring, and Evaluation in the Lower Columbia River and Estuary

Theme	Recommendation	Justification	Report Section	Relevance to CEERP Objective
Fish Health	Derive metrics to assess salmon condition or “fitness.” Develop a non-lethal technique to provide the means to test and predict the impact of various environmental effects on population resilience.	Fish condition near the critical time of ocean entry may set the stage for subsequent survival. A method to measure and compare the fitness of individuals and runs to evaluate and relate to adult returns is needed.	3.4.2.2	May reduce performance in terms of growth and survival
	Conduct experiments on the stock-specific effects of temperature on salmon fitness. Evaluate how climate change may affect spatial and temporal habitat opportunity attributes of restoration projects.	Temperature is a key environmental parameter thought to control salmon physiology, behavior, and fitness. Levels routinely exceed those deemed stressful to salmonids, yet certain genetic stocks persist and can exhibit high growth and high condition factor. A better understanding of the effects of temperature on salmon is needed.	3.2.1.2	May reduce habitat opportunity.
	Map areas of low DO and conduct behavioral/physiological studies.	Low DO has the potential to induce behavioral changes in salmon that may reduce opportunity in poorly flushed habitats and/or increase predation in the lower estuary during ocean advection events.	3.2.1.1	May reduce habitat opportunity and/or performance.
	Ascertain the effects of various contaminants on salmon fitness. Determine pathways of toxic substances to salmon and devise remediation.	Persistent organotoxins are prevalent in juvenile salmon in the lower estuary and may affect salmon fitness at the time they enter the ocean.	3.4.3	May reduce performance in terms of growth and survival. RPA 61.1 and 61.2
Function of Habitats (site scale)	Assess predatory impacts of birds and fish on juvenile salmon in restorations sites. Evaluate possible ecological engineering solutions.	No specific studies of predation on salmon in wetlands have been performed in the LCRE. Bird predation may be especially high in habitats near bird colonies. Yearling predation on fry may also be significant.	3.3.2	May limit capacity of habitats to support salmon. RPA 58.4
	Assess interspecific and intraspecific competitive impacts involving juvenile salmon. Determine possible effects of introduced species.	A limited number of studies investigating competitive interactions have revealed little significant effect. However, additional investigations are warranted, especially regarding introductions.	3.3.3	May limit capacity of habitats to support salmon. RPA 61.1
	Evaluate nutrient and prey fluxes between wetlands and the surrounding environment.	Production and processing of dissolved and particulate matter through wetland environments is largely undetermined but likely to be significant. The extent that prey produced in wetlands but consumed elsewhere is also unquantified.	3.3.1	May increase overall system productivity and salmon capacity. RPA 58.3

Table A.1. (contd)

Theme	Recommendation	Justification	Report Section	Relevance to CEERP Objective
	Initiate experimental studies of ecological interactions between hatchery and naturally produced salmon within selected estuarine habitats, including, for example, effects on salmon feeding behavior and foraging success	Hatchery practices influence size-dependent patterns of salmon habitat use and residency in the estuary. Hatchery and naturally-produced salmon distributions overlap to varying degrees, but the effects of hatchery salmon on the performance of at-risk stocks that are the targets of estuary restoration are unknown.	3.2.5	May limit habitat capacity for naturally produced salmon stocks. RPA 58.1, 58.2, and 61.1.
Ecosystem Function (landscape scale)	Use measurements and models to evaluate levels of connectivity between restoration sites. Metrics can include salmon and dissolved and particulate matter, and should consider spatial and temporal variables.	Linkages between wetland habitats may be a key attribute aiding survival of juvenile salmon, especially fry.	3	May increase overall system productivity and salmon capacity. RPA 59.3
	Evaluate nutrient and prey fluxes between wetlands and the surrounding environment.	Production and processing of dissolved and particulate matter through wetland environments is largely undetermined but likely to be significant. The extent that prey produced in wetlands but consumed elsewhere is also unquantified.		May increase overall system productivity and salmon capacity
	Evaluate the effects of hatchery releases on the estuarine ecosystem. Experimental releases may be necessary to interpret ecosystem responses to hatchery programs.	Hatchery programs account for the majority of salmon produced in the Columbia River basin, and drive salmon abundance patterns, size distributions, and stock composition within the estuary. The effects of concentrated releases of hatchery fish on salmon predators, estuarine food webs, or other ecosystem functions are poorly understood.	3.1.5	Could cause ecological responses that limit the estuary's capacity to support at-risk salmon. RPA 58.1, 58.2, and 61.1.
	Conduct a flow regulation experiment that allows the evaluation of change in opportunity, production/capacity, and realized function of restoration projects for salmon. There is some potential to evaluate variation in these metrics by analyzing data from abnormally wet and dry years as compared to "normal" years.	This experiment would provide guidance regarding the potential effects of water-level management on salmonid habitats and realized function for salmon. Adjustment to flow could have significant effects on access and production/capacity, can could affect habitat formation through hydrologically driven processes.	3.4	Affects both the opportunity to access the sites and production/capacity of the sites. RPA 61.1 and 61.2
	Evaluate the threats from climate change on flows, water-level variation, wetted area of the floodplain, water temperatures, and sea-level rise.	Climate-related factors could strongly affect the quality of restored habitats now and in the future. Although difficult to fully predict, an initial analysis of potential major changes in flow, for example, should be evaluated.	6.4	Affects both the opportunity to access the sites and production/capacity of the sites. RPA 61.4

Table A.1. (contd)

Theme	Recommendation	Justification	Report Section	Relevance to CEERP Objective
Salmon Realized Function				
	Compare juvenile Chinook life histories and performance (i.e., growth, foraging success, survival) among genetic stocks, including less-abundant stocks that are under-represented in past estuarine surveys.	Most lower-estuary collections are dominated by Chinook stocks from the lower Columbia River ESU. Low samples sizes for some ESUs limit understanding of stock-specific habitat use. Additional habitat surveys are needed, particularly in reaches D-H, where some less-abundant stocks (e.g., Willamette River spring, Deschutes River fall, Snake River fall) occur in higher proportions and may reside for extended periods before migrating to the ocean.	4.1.1	Affects abilities to restore habitat opportunities for stocks of interest. RPA 58.2, 58.3, and 61.3
	Establish reference populations in selected tidal tributaries to quantify the estuary's contributions to adult returns based on mark-recapture studies and smolt-to-adult returns. Analyze adult otoliths in other ESUs to determine the relative contribution of estuarine life histories to returning adults.	Restoration programs assume that estuary habitat actions will promote salmon recovery; however, most RME studies only track the performance of individuals within the estuary rather than the estuary's ultimate influence on population success. Selected reference populations representing a diversity of ESUs are needed to quantify the estuary's influence on adult abundance, life-history diversity, and smolt-to-adult returns.	4.1.3	Addresses a fundamental but unproven assumption of CEERP. RPA 58.2, 61.1, and 61.3
	Design restoration "experiments" to directly test population responses to estuary restoration in one or more tidally-influenced tributaries. Ideally, such experiments could incorporate a before-after-control-impact (BACI) design to compare population responses in treated and untreated tidal tributaries.	Restoration effectiveness has been based primarily on measurements of salmon use or performance at the habitat (site) scale. Population-level responses to estuary restoration are poorly understood. Tidally-influenced tributaries may provide a microcosm "estuary" where local population responses to estuarine habitat treatments can be measured.	4.1.3; 5.0	Limits estimation of salmon population responses to restored habitat opportunities
	Use life-cycle modeling to evaluate population responses to alternative estuary restoration actions.	Population-level monitoring is difficult and expensive, and relatively few tributaries may be suitable to directly measure population responses to estuary restoration. Life-cycle modeling offers a useful method for comparing population sensitivities to various survival improvements within the estuary. However, most existing models must be modified to account explicitly for the estuarine phase of salmon life cycle.	4.1.3	Compares population sensitivities to alternative restoration measures. RPA 58.2, 61.1, and 61.3

Table A.1. (contd)

Theme	Recommendation	Justification	Report Section	Relevance to CEERP Objective
	Develop analytical tools needed to further resolve the geographic origins and life histories of individuals sampled in the estuary. Stock identification would benefit from higher resolution genetic baselines. Life history reconstructions would benefit from smaller tags and improved chemical methods for otolith analyses.	The present baseline for Chinook salmon is too coarse to interpret the local stream origins of individuals found in the estuary. Existing PIT and acoustic tags are too large to tag a representative range of all subyearling size classes or life history types. New otolith chemical or structural indicators are needed to distinguish salmon residency in tidal-fresh environments from their residency in natal-stream environments.	4.1.1; 4.1.2	Limits restoration of habitat opportunities necessary to support stock and life history diversity. RPA 58.2, 61.1, and 61.3
	Reconstruct historical salmon life histories and stock abundances to provide context for estuary restoration. Identify restoration actions that can expand life history expression within and among stocks to strengthen population resilience to future disturbance.	Contemporary salmon life histories reflect current habitat opportunities, population structure, and hatchery production practices and may not identify "optimal" targets for recovery. Poorly represented stocks and life history types may indicate more about present opportunities than about restoration potential. A strong historical context is needed to avoid actions that further simplify population structure and reinforce symptoms of stock decline.	3.0	CEERP risks reinforcing a "sliding baseline" of salmon decline if restoration objectives are not placed in historical context

Table A.2. Recommendations for Ecosystem Restoration in the Lower Columbia River and Estuary

Theme	Recommendation	Justification	Report Sections	Relevance to CEERP Objective
AE Monitoring	Incorporate statistical designs into AE research plans.	Integrating analytical plans with AE monitored plans ensures data are being collected in a way that permits appropriate analyses with which to evaluate ecosystem response to restoration actions.	5.0	The quantitative evaluation of restoration will inform changes in opportunity, capacity, and/or realized function of habitats for juvenile salmon. RPA 60.2
	Collect pre-restoration data at reference and treatment sites.	The collection of pre-restoration data facilitates the ability to perform a statistical analyses which permit quantitative evaluations of ecosystem responses resulting from restoration actions. In the absence of pre-restoration data, the ability to surmise meaningful evaluations of restoration activities is severely restricted, and in some, cases may not be possible.	5.0	Evaluate changes in conditions effecting opportunity, capacity, and/or realized function of habitats. RPA 60.1 and 60.2
	Include reference sites in AE study designs.	Reference sites proved a context with which to evaluate monitored metrics at a restored sites, and are necessary for conducting meaningful evaluations regarding the relative successes of restoration projects.	5.0	Evaluate changes in conditions effecting opportunity, capacity, and/or realized function of habitats. RPA 60.1 and 60.2
	Select monitored metrics that are aligned with project goals and objectives as well as those associated with the CEERP program.	Metrics should inform attributes of habitat capacity, opportunity, and realized function. Attributes informing realized function (e.g., juvenile salmon health, growth, and residence time) were the least studied among the AE research projects reviewed. These metrics should be integrated to a greater extent in future AE research efforts to better inform salmon performance within the context of habitat restoration actions in the LCRE.	5.0	Monitored metrics directly related to opportunity, capacity, and/or realized function of habitats for juvenile salmon are critical for meeting CEERP goals and objectives. RPA 60.2
	Combine intensive and extensive monitored locations and metrics at restoration sites throughout the LCRE.	An AE strategy that maximizes spatial and temporal data collection efforts while prioritizing monitored metrics within each study area will create the greatest opportunity for learning from restoration actions in an efficient manner.	5.0	Informs attributes associated with spatial variability of opportunity, capacity, and realized function. RPA 60.2

Table A.2. (contd)

Theme	Recommendation	Justification	Report Sections	Relevance to CEERP Objective
	Evaluate response of juvenile salmon to restoration year-round.	RME research has demonstrated juvenile salmon are present in the LCRE year-round, and yet, AE research has for the most part been focused during spring and summer months. There is a clear need to understand the response of juvenile salmon to restoration actions throughout the year.	5.0	Informs attributes associated with temporal variability on opportunity, capacity, and realized function. RPA 60.2
Tracking	Include standardized report cards for projects to facilitate evaluation of multiple sites across the LCRE.	Provide systematic data set on results of restoration projects toward meeting goals; basis for design of new project; basis for adjusting projects to better meet goals.	5.10	Critical to development of a standard data set on capacity and opportunity. RPA 37 and RPA 60.2
	Develop a centralized database to include monitored data for AE projects.	Allow for access to data for accounting of what was built and analysis of how projects were working for all project types; use for reporting to managers, funding sources, stakeholders, researchers, etc.	5.10	Critical to efficient reporting on improving capacity and opportunity, and for informing selection, design, and implementation of projects. RPA 60
Analysis	Implement a systematic and repeatable method to assess whether there is a net increase, decrease or no detectable change in the LCRE ecosystem, which includes the lowland aquatic habitats as well as the tributary watersheds contributing to the these habitats. Develop a set of indicators of estuarine ecosystem health and ecological integrity, and levels of stress that can be assessed periodically to best characterize the system condition relative to CEERP objectives.	Allow managers and stakeholders to conclude whether actions taken under the CEERP are improving the ecosystem that supports salmonid recovery. It weighs the gains from protection, enhancement, and restoration against the losses from development and other activities. It evaluates whether actions are incrementally reducing stressors to the.	6.6.3; 6.5	Addresses whether there is an increase or decrease in capacity, opportunity, and realized function. Will additionally inform site selection, design, and implementation of projects. RPA 59.5

Table A.2. (contd)

Theme	Recommendation	Justification	Report Sections	Relevance to CEERP Objective
Action	Continue implementing projects that increase the area of functional habitat for juvenile salmon and that provide maximum (~ natural) access to the sites.	The limited research and monitoring of restored sites largely verifies that most of the actions have improved habitat conditions for salmon and that salmon are accessing the sites.	6.3.2	Directly related to all CEERP objectives. RPA 37
	Implement projects that will reduce the exposure of juvenile salmon to contaminants of concern.	Contamination in fish tissue is a concern, and may affect the health of the animals.	3.4.3	Contaminants may reduce the realized function in terms of growth and survival.
	Use new information on wetland plant distribution relative to water level and salinity, and habitat classification in planning projects.	These data provide high resolution and spatially explicit information on the main hydrological condition structuring floodplain wetland assemblages.	6.2.1.2	Directly related to developing the optimal capacity/production of the site.
	Consider, where feasible, incorporating cold-water refuges in project site selection and design.	These refuges may enhance the duration of residence time in restored habitats.	3.2.1.2	Essentially increases the opportunity aspect of the site.